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WATER QUALITY BEHAVIOR IN RESERVOIRS. A COMPILATION OF PUBLISHED RESEARCH PAPERS

James M. Symons

Bureau of Water Hygiene
Cincinnati, Ohio

1969

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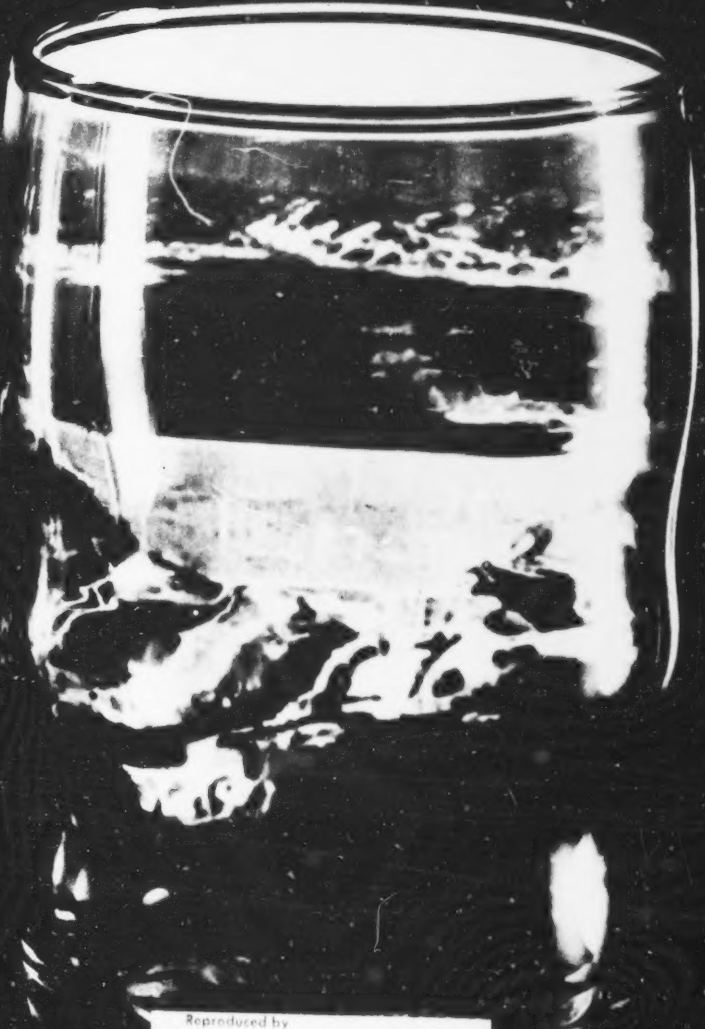

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WATER QUALITY BEHAVIOR IN RESERVIORS

A Compilation of Published Research Papers

Compiled by
James M. Symons

U.S. DEPARTMENT OF HEALTH, EDUCATION, AND WELFARE
Public Health Service
Consumer Protection and Environmental Health Service
Environmental Control Administration
Bureau of Water Hygiene
Cincinnati, Ohio 45213
1969

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Public Health Service Publication No. 1930

FOREWORD

The overall objectives of the project "Influence of Impoundments and Controlled Releases on Water Quality" were to carry out the mandate given in Section 5(d)(1)(C) of Public Law 89-234 (Water Quality Act of 1965), which states, to paraphrase, that an agency of the U.S. Government shall conduct whatever basic and applied research is necessary to permit evaluation of any streamflow regulation project for water quality control proposed by a Federal construction agency.

In this author's nearly 5-year association with this project, many people and several governmental agencies have been involved. To give adequate credit to all persons and agencies who were connected with the research work of this project, the following historical development is presented.

The project was initiated in September 1962 at the Robert A. Taft Sanitary Engineering Center in Cincinnati, and assigned to the Engineering Section, headed by Dr. Richard L. Woodward, in the Basic and Applied Sciences Branch of the Division of Water Supply and Pollution Control, U.S. Public Health Service, Department of Health, Education, and Welfare. At that time the Project Chief was Mr. Samuel R. Weibel.

In November 1963, Dr. Woodward resigned from the Public Health Service and was replaced as Section Chief by Mr. Gordon G. Robeck. At this time, the responsibility for this project was assigned to the author.

The project staff consisted of: Dr. William H. Irwin, Research Aquatic Biologist; Mr. Jack DeMarco, Research Engineer; Dr. Jerzy Kurbiel, Research Engineer; Mr. Ernest L. Robinson, Biologist; Mr. Glenn R. Gruber, Mechanic; and Mrs. Maura M. Lilly, Secretary. In addition to the staff assigned directly to the project, acknowledgment is given to the analytical chemists, supervised by Mr. Jesse M. Cohen, who contributed to and supported this project: Mr. Lawrence J. Kamphake, Mr. Cecil Pinkerton, Mr. George

Holtzer, Mr. Richard T. Shibiya, and Mr. Donald Griffith. Specific acknowledgments appear at the end of each paper.

The present affiliations of the other professional project staff members are as follows: Dr. William H. Irwin and Mr. Ernest L. Robinson, National Water Quality Laboratory, Federal Water Pollution Control Administration, Duluth, Minnesota; Dr. Jerzy Kurbiel, Politechnika Wroclawska, Wroclaw, Poland; and Mr. Jack DeMarco, Bureau of Solid Waste Management, Environmental Control Administration, U.S. Public Health Service.

The project was transferred from the Public Health Service to the Federal Water Pollution Control Administration (FWPCA) in the Department of Health, Education, and Welfare on January 3, 1966. On May 10, 1966, the FWPCA was transferred from the Department of Health, Education, and Welfare to the Department of the Interior. In January 1967, the project was transferred to the FWPCA laboratory at the Robert S. Kerr Water Research Center in Ada, Oklahoma.

The author will continue research work on the public health aspects of surface water protection in the Division of Criteria and Standards, Bureau of Water Hygiene, Environmental Control Administration, Consumer Protection and Environmental Health Service, U.S. Public Health Service, in Cincinnati, Ohio.

Finally, the author would like to thank those who helped with the preparation of this Compilation. They are: Mr. J. Keith Carswell, Surface Water Protection Section, Bureau of Water Hygiene, and the author's wife, Joan, who proofread the material; Mrs. Florence B. Layne, who edited the manuscript; Mrs. Marion G. Curry, who edited the references and prepared the index; Mrs. Pauline J. Elliott, who composed the camera copy; Mr. Newell J. Mastin, who prepared the figures; and Mr. Dale W. Dietrich, who designed the cover and supervised the makeup. The latter three are on the staff of the Office of Public Information and Education, Environmental Control Administration, and Mrs. Curry and Mrs. Layne were formerly with that office.

James M. Symons
Chief, Surface Water
Protection Section

Cincinnati, Ohio
July 1969

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ABSTRACT

This volume contains all the technical papers prepared from data collected for the project "Influence of Impoundments and Controlled Releases on Water Quality," from September 1962 through December 1966. Research on this project was divided into two broad categories: 1) investigative research, designed to better understand reservoir water quality behavior; and 2) engineering research, designed to develop engineering methods for the control of reservoir water quality. Papers included in this Compilation cover a review of the literature on the project topic and a statement of research needs; and research on the dissolved oxygen budget in reservoirs, on the behavior of the nitrogen cycle in reservoirs, on the behavior of synthetic organics in stratified reservoirs, and on the use of artificial destratification as an engineering technique for managing and maintaining good quality water in reservoirs.

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INTRODUCTION

This volume contains all the technical papers prepared from data collected for the project ''Influence of Impoundments and Controlled Releases on Water Quality,'' from September 1962 through December 1966. Although technical data in the general project area will continue to be developed by the staff at the Robert S. Kerr Water Research Center, Federal Water Pollution Control Administration, the data will be different in scope and emphasis from those developed by the staff located in Cincinnati. Therefore, compilation of their research papers up to 1969 is appropriate as a summary of the first phase of this long-range research project.

Each of the papers is reproduced essentially as it appeared in the technical literature with only minor editing and updating. Correct reference citations are given in the introduction to each paper. For the papers presented in the *Journal of the Sanitary Engineering Division, American Society of Civil Engineers*, the discussions that appeared following original publication and the ''Authors' Closure'' are also included. Three papers contain transcriptions of the floor discussions that followed presentation of the papers. These papers originally appeared in various forms – in technical journals, in conference proceedings, and as reports, and appeared out of proper sequence. This Compilation, with 1) the material re-edited, updated, and corrected; 2) the material in a single document and in proper sequence; 3) additional text added in areas that have proven weak; 4) inclusion of written and oral discussions following the appropriate papers; and 5) inclusion of an index, should prove useful to those investigators interested in water quality in lakes and reservoirs. Individual reprints are still obtainable for most of the papers.

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Research on this project was divided into two broad categories: 1) investigative research, designed to better understand reservoir water quality behavior; and 2) engineering research, designed to develop engineering methods for the control of reservoir water quality. Seventeen papers originally were prepared on different phases of this project. Papers 1 and 2 are, respectively, a review of the literature on the project topic and a statement of research needs, and a summary of the overall research plan with some early results.

In the investigative research category, paper 3 and paper 4, part 1, contain the data collected in the studies concerning measurement of the dissolved oxygen budget in reservoirs; papers 5 and 6 contain information on the behavior of the nitrogen cycle and of synthetic organics in impounded waters. In the engineering research category, papers 7, 8, 9 (paper 4 - part 2), 10, 11, 12, 13, and 14 contain results on the use of artificial destratification as an engineering technique for managing and maintaining good quality water in reservoirs.

In addition, three summary papers were prepared for specific audiences. Paper 15 summarizes early research information, paper 16 was prepared for the International Association of Scientific Hydrology, and paper 17, which is the project report, combines a paper for the 17th Annual Sanitary Engineering Conference at the University of Kansas and a paper for a Seminar on Reservoir Destratification for the Department of Water Resources, State of California.

Additional copies of this volume may be obtained at no cost by writing to: Dr. James M. Symons, Chief, Surface Water Protection Section, Bureau of Water Hygiene, Environmental Control Administration, U.S. Public Health Service, 5555 Ridge Avenue, Cincinnati, Ohio 45213.

PART I

LITERATURE REVIEW
AND
OVERALL
RESEARCH PLAN

I

PAPER NO. 1

INFLUENCE OF IMPOUNDMENTS ON WATER QUALITY

A Review of Literature and Statement of Research Needs

J.M. Symons, S.R. Weibel, and G.G. Robeck

The current and pertinent literature on a wide variety of subjects relating to reservoir water quality is here summarized in one document. At the end of each subsection, some of the gaps in knowledge and suggested research on that topic are noted. Although this summary has not been revised since January 1966, and several important papers have been published since then, it still gives a good overall view of the subject area.

The correct reference citation is:

Symons, J.M., S.R. Weibel, and G.G. Robeck, "Influence of Impoundments on Water Quality - A Review of Literature and Statement of Research Needs." PHS Publ. No. 999-WP-18, October 1964. Revised January 1966. 78 pp.

FOREWORD

The enactment on July 20, 1961, of Public Law 87-88, an amendment to the Federal Water Pollution Control Act (P.L. 660), caused the Federal construction agencies, e.g., the Corps of Engineers and the Bureau of Reclamation, to begin giving consideration to the inclusion of storage for regulation of streamflow for the purpose of water quality control in the survey or planning of any impoundment. Since, according to P.L. 87-88, the

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need for and the value of storage for this purpose shall be determined by these agencies with the advice of the Secretary of the Department of Health, Education, and Welfare, the act instructs the appropriate group in the Department to do whatever basic and applied research is necessary to permit evaluation of the effects on water quality and water uses of regulated streamflow to control water quality. The Engineering Section of the Basic and Applied Sciences Branch of the Division of Water Supply and Pollution Control at the Robert A. Taft Sanitary Engineering Center is the group making these studies.

Research on the influence of impoundments is a portion of a project entitled ''The Effects of Streamflow Regulation on Water Quality.'' Since any streamflow regulation project having the goal of water quality control involves the release of impounded waters at times of deteriorating downstream water quality, the influence of impoundments on water quality must be known if the project is to be successful. A discussion of this phase of streamflow regulation is the major concern of this report.

Two other aspects of streamflow regulation are so closely associated with impoundment behavior that they are also included in this report, although to a lesser extent. One is the influence of impoundment releases on downstream water quality, and the other is the use of the systems analysis approach correlated with economic analysis for the optimization of large water quality control projects.

This report is broad by design, touching on the fields of chemistry, microbiology, algology, physical phenomenon, measurement technology, stream self-purification, economics, and systems analysis, since one of the important purposes of the report is to indicate to readers who are new to the field of impoundment behavior and streamflow regulation the enormous breadth of these fields. The other two purposes are: (1) to discuss each topic in sufficient detail to give the reader insight into the current understanding of the topic and (2) to indicate the major research needs in each problem area and to suggest possible fruitful avenues of study to satisfy these needs.

To achieve these purposes, an intensive review was made of over 600 literature references. The 149 most pertinent were then selected as the basis for this report. Since this report summarizes current thinking on each topic, the selected references for the most part are current ones. About 80 percent of them were published from 1958 to 1964. A reader wishing to delve further into a given topic will find the bibliographies in the documents referenced most useful in leading him back through the literature.

Since an important objective of this report is to indicate the research needs in various areas, the gaps and insufficiencies in current knowledge are pointed out in each subsection. This was done, not in an effort to cast doubt on the value of the work by others, but to indicate that in most areas relating to impoundment behavior more work needs to be done before the subject is completely understood. The sections devoted to research needs are often stated in general terms because a need has been recognized, but a specific and detailed plan of attack has not yet been formalized. In this orienting and summarizing report, this approach seems perfectly proper.

In summary, this report attempts to indicate: (1) the broad scope of influences that impoundments have on water quality, (2) the existing information connected with various phases of these influences, and (3) some of the most pressing research needs in this area. This report can serve to guide research, not only within the U.S. Public Health Service, but also in universities, private research organizations, and other governmental agencies.

The authors wish to thank the following persons for their helpful and critical reviews: Mr. Bernard B. Berger, Mr. Frank E. DeMartini, Mr. Frank H. Rainwater, Dr. William M. Ingram, Mr. Kenneth M. Mackenthun, Mr. John E. McLean, Mr. Robert W. Haywood, Jr., and Dr. Richard L. Woodward.

INTRODUCTION

The initiation of research by the U.S. Department of Health, Education, and Welfare on the influence of impoundments on water quality was stimulated by Section 4(d)(1) of Public Law 660 as amended by P.L. 87-88, enacted July 20, 1961, which states in part:

> "Section 4(d)(1) In carrying out the provisions of this section the Secretary shall develop and demonstrate under varied conditions (including

> conducting such basic and applied research, studies, and experiments as may be necessary):
>
> ''(C) Methods and procedures for evaluating the effects on water quality and water uses of augmented streamflows to control water pollution not susceptible to other means of abatement.''

and by Section 2(b) (1 and 2), which states:

> ''(b)(1) In the survey or planning of any reservoir by the Corps of Engineers, Bureau of Reclamation, or other Federal agency, consideration shall be given to inclusion of storage for regulation of streamflow for the purpose of water quality control, except that any such storage and water releases shall not be provided as a substitute for adequate treatment or other methods of controlling waste at the source.
>
> ''(2) The need for and the value of storage for this purpose shall be determined by these agencies, with the advice of the Secretary, and his views on these matters shall be set forth in any report or presentation to the Congress proposing authorization or construction of any reservoir including such storage.''

This very large research mandate, under the general project title ''The Effects of Streamflow Regulation on Water Quality,'' has been initially divided into five parts for study:

1. Characterization and treatment methods for stormwater runoff from urban areas.

2. Characterization and treatment methods for stormwater runoff from rural areas.

3. Investigation and reevaluation of stream pollution criteria.

4. Determination and prediction of the behavior of water quality in flowing streams.

5. Determination and prediction of the behavior of water quality during impoundment.

The discharge of untreated or partially treated stormwater runoff from urban and rural areas is important since it represents one source of ''water pollution not susceptible to other means of abatement,'' to quote amended P.L. 660. Studies on urban runoff have been summarized by Weibel, Anderson, and Woodward,[1] and reviews similar to this one are being prepared on urban and rural runoff. Stream pollution criteria and the behavior of water quality in flowing streams are important since stream pollution in all forms must be recognizable and the influence of dilution water on stream behavior must be predictable. Finally, the changes that occur in water quality during storage in a given environment must be understood and predictable since impounded water is the source of water for regulating streamflow. If a streamflow regulation project is to be successful, the capability to forecast the quality of the water available for dilution must be developed.

The literature on the behavior of water quality during impoundment was extensively reviewed for this report. This review was then summarized so that the current state of information relating to various phases of the problem could be reported and the future research needs delineated. Although the major emphasis in this report is on the problem of the determination and prediction of the behavior of water quality during impoundment, the closely related problem of the influence of impoundment releases on downstream water quality and the use of systems analysis methods coupled with economic analyses for streamflow regulation project design are also discussed. The legal aspects of streamflow regulation for water quality improvement, such as the problem of water rights, are not, however, covered.

In reviewing the literature on the topics of stream and impoundment behavior, one is immediately struck by the vastness of the writings on these subjects. This is reasonable, however, since these two topics encompass almost the entire sciences of sanitary engineering and limnology, both of which are broad fields of study. Most articles are a detailed description of some particular research that adds new knowledge in a specific area of interest, but some works, such as ''Water Pollution -- Economic Aspects and Research Needs,'' by Allen V. Kneese,[2] discuss the overall problem, as do the reports to the Select Committee on National Water Resources, United States Senate.[3] An example of research covering the broad problem is the U.S. Public Health Service Research and Training Grants Branch's 6 years of sponsored research (Grant number WP-00153) at the Johns Hopkins University under the project title ''Low-Flow Augmentation for Stream-Pollution

Abatement."[4] (References 58, 71, 90 are other reports pertaining to this same project.)

With so much literature on the subject, one might wonder whether problems could still exist; but many do. In spite of all this research, sufficient information is not available to permit the choice of one impoundment site from among several in a given geographical area that will have the minimum adverse effect on the quality of the water and that will, therefore, provide the best quality water. Further, if an impoundment site is fixed, sufficient information is not available to permit prediction of how the incoming water will be changed as it stands in this impoundment. Finally, because of this lack of understanding, firm recommendations cannot be made relating to outlet structure design, impoundment operating practices, or in-impoundment control, since the influence of changes in these items on discharged water quality is not known.

Why, with so much research in the appropriate general area, does uncertainty still exist in the ability to predict the influence of impoundments on water quality? There are at least three reasons. First, much of the data, though generally applicable to the understanding of impoundment behavior, has not been taken with impoundment behavior specifically in the researcher's mind. For example, extensive research by limnologists and aquatic biologists has related the growth of algae to light intensity and penetration. Little attention has, however, been paid to the increased light penetration that is caused by impounding a turbid river water and to the potential organism growth problem that this clarification may pose.

Second, many of the investigations have not been fundamental enough to allow the observed phenomena to be projected into a new set of environmental conditions. For example, for years thermal stratification has been noted in impoundments, yet the necessary set of environmental circumstances that enhance the possibility of thermal stratification has never been completely delineated.

Third, the environmental conditions in an impoundment differ from those in a flowing stream in many ways, e.g., changes in water depth, increased detention time, and possibility of thermal stratification. Each influences the normal biological, physical, and chemical processes that cause changes in water quality.

Many of these processes have been studied under flowing stream conditions, but have not been investigated in impoundment environments where the additional detention time allows natural processes to proceed far beyond the extent possible in a flowing stream. In addition, if thermal stratification exists in an impoundment, the fact that the bottom portion of the water is trapped and does not contact the atmosphere or aerated water has a marked effect on the natural processes that occur in the water.

For impoundment environments, therefore, it is necessary to: (1) investigate new relationships thoroughly, (2) reinvestigate known relationships, and (3) relate previously observed results to the environmental factor or factors that caused them so that in a different environment different results can be predicted. Until these three important general research needs in the area of impoundment behavior are met, streamflow regulation projects for quality control will not be founded on a firm scientific basis. Despite the predictions of impoundment and stream behavior currently being made, there is need for greater knowledge.

The report is divided into three major sections: (1) the behavior of water in impoundments, (2) the influence of impoundment releases on downstream water quality, and (3) operations research for water quality management. Each major heading is subdivided for clarity; each subsection contains sections on ''Existing Information'' and ''Research Needs.'' A summary of current streamflow regulation projects and future research activities is also included.

BEHAVIOR OF WATER IN IMPOUNDMENTS

Measurement of Water Quality in Impoundments

HAND SAMPLING

Existing Information

The effect of impoundments on water quality and the measurement of water quality in impoundments have been under investigation for many years. The earliest reference found is an article published in 1890 by Stearns,[5] ''Effect of Storage on the Quality of Water.'' Since that time, and continuing to the present with projects such as the study of the water quality in Lake Michigan under the auspices of the Great Lakes - Illinois River Basin project, data have been taken and articles written concerning the quality of water in impoundments.

This review developed 80 references to water quality studies in specific impoundment locations, and the list is not exhaustive. In spite of this wealth of information, the generalized behavior of water quality in impoundments is not completely understood, mainly because of the method by which the data in these individual studies were taken. Often, these data represent an isolated set or a series of sets of hand samples that were analyzed for the common chemical and biological water quality characteristics. In general, no attempt was made to: (1) measure the changes that occurred in water quality with time, or (2) relate the magnitude of any given change to the environmental condition influencing it. While these studies served the purpose for which they were intended, in the longer view they consisted more of ''data taking'' than of research investigations. This is often true of field investigations. Two reports that attempt to avoid the difficulty mentioned will be noted in detail later.

In addition to numerous reports on the measurement of the quality of water in lakes and impoundments, several articles sought to discuss in qualitative terms the influence of impoundments on water quality. One article by Churchill,[6] ''Effects of Storage Impoundments on Water Quality,'' covers most of the generalized effects of impoundments on water quality. The data taken by the Tennessee Valley Authority during 20 years of operation are discussed in detail with emphasis on the following:

1. Development of thermal stratification and its effects.
2. Reduction in turbidity attributed to the storage capacity of reservoirs.
3. Reduction in color concentration attributed either to biological activity, sunlight, or the equalizing action of the impoundment.
4. Possible production of H_2S odor or algal odor.
5. Reduction in coliform organisms.
6. Effect of impoundments on BOD and dissolved oxygen in the water.
7. Buildup of free carbon dioxide in the bottom waters of stratified reservoirs.
8. Possible nitrification and denitrification in impoundments.
9. Possible dissolution of iron and manganese in bottom waters.
10. General equalization or smoothing effects of a storage impoundment on mineral quality.

The article also discusses reaeration of deoxygenated water below dams. In spite of the usefulness and thoroughness of this article, the author's remark that, ''The facts obtained and the general principles deduced therefrom should be helpful in forecasting the changes in water quality that are likely to occur in any proposed reservoir,'' seems premature.

An article by Love,[7] ''Relationship of Impoundments to Water Quality,'' discusses similar subjects with the addition of density flows in impoundments, heat dissipation in impoundments, and behavior of radioactivity in impoundments. His conclusion differs somewhat, however, from Churchill's in that he states:

> ''Many studies of stream impoundments have been made, and much valuable information has been obtained. Unfortunately, most of the studies have been more qualitative than quantitative. Even when water characteristics have been reported quantitatively, overall evaluations of water storage projects and their relation to multipurpose interests have been largely qualitative. There is an increasing need for more thorough studies of stream impoundments in terms of parameters usually passed over lightly or completely ignored.''

At least two reports have attempted to measure water quality as it changes with time. One, ''Mainstem Reservoir Effects on Water Quality in the Central Missouri River,'' by Neel, Nicholson, and Hirsch,[8] measures the effects of six mainstem reservoirs on the quality of water in the Missouri River over the period 1952 to 1957. Although all the environmental conditions that cause changes in water quality are not given, several explanations for the observed effects are given. These discussions help make the data in this report useful for understanding the generalized behavior of impoundments. The noted effects cannot be applied indiscriminately, however, since these reservoirs do not thermally stratify and exhibit plug flow with a minimum of longitudinal mixing, which may be the case in most run-of-river impoundments. In spite of these limitations, this study is valuable.

A similar study was made from March 1945 to March 1946 by the Tennessee Valley Authority. During this time 89 samples taken above and below Cherokee Reservoir were analyzed for common water quality characteristics. These included: temperature, dissolved oxygen, BOD, color, pH, alkalinity, CO_2, all four nitrogen forms, turbidity, solids, phosphorus, manganese, and chlorides. In addition, coliform organism densities were determined. Although no environmental conditions were noted, these data give an excellent picture of the change in water quality as water passes through a stratified impoundment.

Because of the stratification and resulting density flows, the holdup time for any given influent water varied throughout the year. In the fall of the year the cool influent water flowed directly along the bottom under the warmer reservoir water and out the penstocks in as little as 2-1/2 weeks. During the spring and summer the detention time was found to be as long as 25 weeks because the warm influent water spread across the surface of the impoundment and was slowly drawn down to the outlet level to operate the power turbines. The warm influent never appeared as a distinct volume of water. In view of the different behavior of the warm and cold influents, any comparison of influent and effluent data from this study should be considered with caution. The Cherokee Reservoir study is, however, the most complete evaluation of the changes in water quality in a stratified impoundment found in the literature search. The data have not been published, however, and were obtained through a private communication from Milo Churchill, Chief of the Stream Sanitation Staff of the TVA.

Research Needs

Field research investigations need to be made, and the data taken should be correlated with the environment. This is important since the great mass of data in the literature does not permit generalization because of variations in impoundment geometry, soil conditions, environment conditions, and so forth.

Environmental factors that should be reported are: character of soils in bottom and bank storage areas; length, width, and depth ratios of the reservoir; reservoir orientation; wind direction and velocities; solar radiation intensity; and light penetration. These environmental data, correlated with adequate numbers of samples taken over an extended period of time, should allow a relationship to be drawn between environmental conditions and the changes in water quality. Care must be taken in the definition of significant factors and the design of the experiment since a mass of data is of limited use if it cannot be interpreted in such a manner as to permit the emergence of a general law, assuming one exists.

CONTINUOUS MONITORING

Existing Information

The development of instrumentation for continuous monitoring of water quality is just beginning. Several continuously recording monitors that make a permanent record of such characteristics as dissolved-oxygen concentration, pH, conductivity, temperature, turbidity, solar radiation intensity, and air temperature are currently commercially available. Water levels and other weather information can also be recorded, if required. An article in *Chemical and Engineering News*, ''Monitor System Continuously Records River Water Quality,''[9] describes a continuous monitoring system operating on the Delaware and Schuylkill Rivers. Ragone and Peters[10] in a paper entitled ''Water Quality Monitoring for Water Quality Control,'' describe a continuous monitoring system that the Virginia Electric and Power Company has on the Roanoke River in North Carolina. Continuous monitoring is important to the power company since its Federal license contains a specification for minimum allowable dissolved-oxygen concentration downstream from the dam They must therefore, continuously monitor the dissolved-oxygen concentration downstream from their power dams to have continuous proof of compliance with the provisions of the Federal license. Cleary[11] in his article ''An Electronic Monitor System for River-Quality Surveillance and Research,'' describes the continuous monitor developed for ORSANCO.

Research Needs

The research needs in this area fall into four categories: (1) testing and improvement of available continuous monitors, (2) development of methods of continuously monitoring other water quality characteristics, (3) adaptation of existing monitors designed for use in flowing water to impoundment studies, and (4) collection of significant amounts of continuous quality data in existing impoundments. The Basic Data Branch of the U.S. Public Health Service has done much in developing and evaluating monitors. Their current results are summarized in ''How Automation Plays a Part in Water-Quality Surveillance'' by McCallum and Stierli[12] and ''Instrumentation in Water Quality'' by Mentink.[13]

Although field data must be carefully taken to avoid some of the criticisms mentioned above, there can be no doubt that extensive field data must be taken to confirm the applicability of laboratory experiments to prototype conditions. These data can probably be best taken by continuous robot monitors, which reduce the manpower necessary to take the data and provide a continuous record of water quality rather than an intermittent record resulting from periodic grab samples.

Data should also be taken throughout the depth of an impoundment. Immersing the sensing elements to the appropriate depth seems a better scheme than pumping the sample to the surface for monitoring since the latter procedure may cause changes in the quality characteristics to be measured. The development of completely submersible sensing units is a logical alternative and presents another research need.

Development of Organisms

EUTROPHICATION

Existing Information

Limnologists, aquatic biologists, and sanitary engineers have shown increasing concern in recent years about the continual tendency toward eutrophication of lakes and impoundments. Some recent studies and reports on Lake Geneva,[14] Lake Constance,[15] and Lake Tahoe[16] are examples of publications expressing concern about the condition of large lakes. The one major concern about these lakes is the increasing potential for the development of nuisance organisms, mainly algae, because of the addition of domestic wastes. These wastes tend to supply the nutrients for algal growth. Their increasing concentration presents, therefore, a potential hazard with respect to the degradation of the lake through the excessive growth of nuisance algae.

Much research effort has been directed by various investigators toward determining the nutrient requirements of various species of algae. Much of this research has tried to determine the minimum nitrogen and phosphorus concentrations that would permit the growth of algae.

Widely quoted figures are given by Sawyer[17] in ''Fertilization of Lakes by Agricultural and Urban Drainage.'' A portion of this important reference is quoted below.

> ''All the evidence obtained in the Madison survey lends support to the belief that phosphorus is a key element in determining the biological activity in a body of water. The data regarding critical levels of organic phosphorus are not too well correlated but, in the main, show that most of the lakes producing nuisance blooms have average concentrations in excess of 0.10 ppm [100 μg/l]. The data concerning critical levels of inorganic phosphorus, however, are much more decisive and indicate that nuisance conditions can be expected when the concentration of inorganic phosphorus exceeds or equals 0.01 ppm [10 μg/l].
>
> ''The importance of nitrogen in supporting algal blooms cannot be denied. A critical level of 0.30 ppm [300 μg/l] of inorganic forms is indicated from the data obtained. However, it should be pointed out that extensive algal growths were produced under laboratory conditions with plentiful supplies of phosphorus and deficient supplies of nitrogen. Undoubtedly, nitrogen fixation, either bacterial or algal, bridges the deficiency and produces the nitrogen necessary for the synthesis of algal protein. In the absence of plentiful supplies of phosphorus, however, nitrogen fixation was found to be unimportant.''

In contrast, Hutchinson[18] in *A Treatise on Limnology*, Volume I, reports nitrogen values of from 50 to 85 micrograms per liter in three Wisconsin lakes that produced regular nuisance blooms of algae. Gerloff and Skoog[19] in ''Nitrogen as a Limiting Factor for the Growth of *Microcystis aeruginosa* in Southern Wisconsin Lakes'' found that an N/P ratio of 60/1 is optimal to produce growth, although the N/P ratio as it occurs in algae is 10/1. This reduction is confirmed by Strickland[20] who states, in ''Measuring the Production of Marine Phytoplankton,'' that if nitrogen is present in excess, algae can develop with a phosphorus content of only 20 percent of its normal value. The usual phosphorus content of algae averages 1 percent of the dry cell weight. The ability of algae to grow in the presence of suboptimal concentrations of nitrogen and phosphorus complicates the problem. The nutrient requirements in an impoundment environment with respect to nitrogen and phosphorus are further complicated by the recycling of nutrients caused by death and disintegration of algae. This is reported by Mackenthun[21] in ''The Effects of Nutrients on Photosynthetic Oxygen Production in Lakes and Reservoirs.'' Another complicating factor is the nitrogen fixation ability of many algae, which is reported by Sawyer, Ferullo, Dugdale, and Neess in ''Algae and Metropolitan Wastes''[22] and by Sawyer.[17] The varying nitrogen and phosphorus requirements of different species of algae are also a complicating factor.

Finally, there is the problem of other nutrients that these organisms require. Lackey[22] states, ''When we note the condition of blooms in the field, it is at once apparent that there is much variation of the conditions under which they appear, and at best, our knowledge of what nutrient substances and amounts are related to algal densities, is merely a guess.'' Later, referring to *Euglenophyceae*, he states, ''However, for these last, it is highly probable that undefined extractives and dissolved organic substances (often from the excreta and dung of animals) are also necessary.'' Finally, he remarks, ''There has been considerable discussion of the possibility of removing PO_4 from the effluents of conventional treatment plants. It is suggested here that this may change only the composition of the bloom. There are algae whose PO_4 requirements are extremely low, below the concentration which might remain after treatment.''

In the same report,[22] Provasoli discusses the vitamin requirements of algae. In a table listing 154 species of algae, he notes that 98 have a vitamin requirement. He further states, ''The above data clearly indicate that B_{12}, Thiamine, Biotin, and perhaps other unknown growth factors are ecological factors which we cannot afford to neglect.'' Extensive marine ecology literature notes vitamin B_{12} as an algal growth stimulant. Rodhe,[23] reporting on the inorganic requirements of algae in ''Environmental Requirements of Fresh Water Plankton,'' states that iron, magnesium, and potassium all influence the growth and development of algae in a manner similar to that exercised by nitrogen and phosphorus. Goldman[24] found molybdenum a limiting factor in Castle Lake, California, and one or more trace elements limiting in the lakes of Alaska and New Zealand. In addition to these requirements, diatoms have a specific requirement for silica.

In a personal communication, Mackenthun of the U.S. Public Health Service stated that the situation in Lake Wingra, in Wisconsin, is one of ample nitrogen and phosphorus concentrations. In spite of this, a balanced and controlled algal growth, rather than a nuisance bloom, has developed. The possibility of a growth factor being missing in this situation cannot be overlooked. In a personal communication, Maloney of the U.S. Public Health Service said that the precipitation of iron (as iron sulfide), because of the formation of hydrogen sulfide in lake waters, prevents continued algal growth owing to a lack of iron. Iron must, therefore, be another trace nutrient. These five reports indicate how truly complex the nutrient requirement problem is. Although considerable research work has been done, this problem is not yet completely understood.

It must be remembered that algae provide an important link in the overall food chain. Difficulties arise only when these organisms become extremely out of balance, and a bloom occurs. These difficulties are well summarized in a paper by Mackenthun;[21] pertinent extracts are given in the following quotation:

> ''Enrichment often results in domination of the algal mass by a relatively small group of blue-green algae which becomes well established. ... Fish kills have resulted from supersaturation of oxygen, as well as depletion of oxygen. ... When algal cells die and sink to the hypolimnion, oxygen is used in their decomposition. ... Fishing may be impaired. Bathing, boating, and water skiing often are indefinitely postponed in waters which otherwise would offer maximum multiple recreational use. Industrial and municipal water treatment is hampered and made ineffective by extensive aquatic growth; property values are lowered, and often resort trade is cancelled as a result of these nuisances. Rapid decomposition of dense algal scums with associate organisms and debris gives rise to odors and hydrogen sulfide gas, which creates strong citizen disapproval and often stains the white paint on residences adjacent to the shore.''

Another useful summary of the overall problem is contained in ''Stream Pollution by Algal Nutrients'' by Stumm and Morgan.[25]

Hasler[26] is another investigator who has written extensively on this subject. In his article "Eutrophication of Lakes by Domestic Drainage," he summarizes the problem by citing Lake Zürich in Switzerland. This lake is composed of two basins separated by a narrow passage. Less than 100 years ago both basins were clear and clean. About 1895, however, the lower basin began to receive urban drainage from several towns with a total population of about 110,000. The City of Zürich does not contribute wastes to the lake, although it is along the shores.

Shortly thereafter, problems developed. To quote Hasler:

> "Hand in hand with the increased urban fertilization there developed (beginning in 1896) algal scums, malodorous and displeasing to the eye. The lake ceased to be an attractive site for general vacationing, boating, and bathing. Moreover, since the cities and their industries got their water from the lake, expensive filtering and purifying systems had to be installed to remove the greatly increased mass of organisms and their accompanying obnoxious odors and flavors. ... The shallower of the two lakes, the Obersee, which receives no urban drainage, changed little and still retains its 'virgin' lake characteristics."

The problems that developed in the lower basin of Lake Zürich constitute a classic example of accelerated eutrophication because the Obersee remained clear and serves as a control for comparison.

Research Needs

For a better understanding of the relationship between eutrophication in natural waters and development of algae, the true nutrient requirements of some of the more common species of problem algae must be established. The nutrients might be determined by growing algae in a simple well-defined medium in which growth factors and inorganic trace elements could be varied independently. Through such a systematic investigation, the true nutrient requirements of the various algal species could be evaluated. These laboratory data should then be taken to field situations, such as Lake Wingra, to demonstrate an understanding of the cause of the unexpected behavior.

Many competent investigators are well aware of the needs in this area and have been working toward a complete definition of these nutrient requirements. Examples of investigators in this area are a group of General Dynamics Corporation scientists, Kettering Laboratory scientists, and Drs. Skoog and Gerloff at the University of Wisconsin. The occasional unexpected behavior that occurs in the field indicates, however, that all facets of algal nutrition are not yet known.

Two other factors must be considered as part of the eutrophication problem, algae-destroying viruses and algal toxins. Algal viruses have been isolated, and their algae-destroying properties are described by Safferman and Morris[27] in ''Control of Algae with Viruses.'' The existence of algal toxins is thus far speculative, but in the late fall of 1963 the algal counts in the Markland Pool of the Ohio River at Cincinnati were well below the expected level, in view of the quality of the water at that time. One possible cause of this could have been algal toxins.

Two recent articles discuss techniques that may prove useful in the suggested studies on the effects of nutrients and toxins on the development of organisms. Goldman[28] in ''A Method of Studying Nutrient Limiting Factors *In Situ* in Water Columns Isolated by Polyethylene Film'' describes a static test in which a long, large isolated column of water was inoculated with a given material and the influence of that substance on the water column determined by comparison with the behavior of the external water. In their paper ''Further Measurements of Primary Production Using a Large-Volume Plastic Sphere,'' Antia, McAllister, Parsons, Stephens, and Strickland[29] describe a similar technique, except that their apparatus is free floating and continually mixed, because of rotation. Both of these techniques are interesting and potentially helpful.

LIGHT PENETRATION AND TURBIDITY REMOVAL

Existing Information

Limnologists have given considerable attention to the effects of light intensity on the development and growth of algae. Varma and Wilcomb,[30] in an article entitled ''Effect of Light Intensity on Photosynthesis,'' show a direct relationship between light intensities and photosynthetic rates. Optimum light intensities are not, however, necessary. Verduin[31] points out in a discussion of ''Photosynthesis As a Factor in the Oxygen Balance of Reservoirs'' that less than 2 percent of the optimum light intensity supports a healthy algal bloom. Light intensities above the optimum are actually detrimental to algae, as Vaccaro and Ryther[32] point out in their article ''The Bactericidal Effects of Sunlight in Relation to 'Light' and 'Dark' Bottle Photosynthesis Experiments.'' They also state that if water is brought from some depth to the surface, placed in containers, and returned to the original depth, the algae may be injured by being subjected to a high light intensity during the trip to the surface. This also has an influence on the behavior of algae that are being mixed vertically. This type of mixing results in the continual movement of the algae from positions of low light intensity to those of high light intensity.

Metzler's[33] paper ''Improvement of Water Quality in Kansas'' was the only one found that specifically notes that the decrease in turbidity in impoundments caused increasing productivity, but he reports this only in a qualitative manner. Churchill of TVA, as noted in a private communication, and Neel, Nicholson, and Hirsch[8] measured the reduction in turbidity caused by impounding waters. In the TVA's Cherokee Reservoir, reductions in turbidities from 40 to 60 Jackson units in the influent to 10 or less in the effluent were quite common throughout the year-long study. In the Fort Randall Reservoir on the Missouri River, reductions in turbidities from 400 to 500 Jackson units to less than 35 were measured over a 6-month period. All other conditions being the same, this clarification of the river waters causes potential problems. Dvihally-Tamas,[34] in ''Transformation of Energy in Inland Waters Examined on the Basis of Investigations Carried out Using Modern Physico-chemical Methods,'' relates the penetration of certain wavelengths of light in different quality waters to the productivity level of the water. In one case, waters with relatively large sodium concentrations allowed the penetration of red light, the most favorable for photosynthesis, whereas in the waters of another river the wavelengths of light that reached the deeper layers were ineffective from the point of view of photosynthesis. He concludes, therefore, that, ''The transformation of energy therefore depends on the turbidity of the water, adsorption of light, and spectrocomposition.''

Research Needs

The prime research need is to relate quantitatively the reduction of turbidity and the increased penetration of light caused by the relatively long detention times in impoundments to the increased productivity of these bodies of water, since this is potentially a serious problem. Although this relationship is noted by Metzler,[33] it has never been specifically measured in, for example, a newly created impoundment. This needs to be investigated both with laboratory and field studies. The Pearl River Impoundment Project[35] is to measure water quality in a newly created impoundment before (river water quality) and after (impounded water quality) construction of the dam. These data may help satisfy this need. Further attention should also be given to the practicability of reducing productivity by blocking light with coverings such as powdered carbon or floating polystyrene.

OXYGEN PRODUCTION AND DEMAND

Existing Information

The measurement of oxygen production by plankton has been of considerable interest to both limnologists and sanitary engineers, although limnologists are generally more interested in primary production of new organism mass than in oxygen production. Many articles in the limnologists' literature discuss methods of measuring primary production and methods for eliminating the complicating feature of simultaneous oxygen production and organism respiration. When dealing with an impoundment, the sanitary engineer is interested in the net effect of the algal population in the oxygen balance, rather than the gross production of oxygen by the algae. A study on the Truckee River by O'Connell, Thomas, Godsil, and Hirth[36] is a good example of an attempt to measure the net rather than the gross influence of plankton on oxygen concentrations.

Verduin[31] summarizes much of the limnological literature on this topic and suggests that the gross photosynthetic rate for an average algal density is 16 grams of oxygen per square meter per day. For a 10-meter euphotic zone (the depth in which 99 percent of the incident light is absorbed), this amounts to 1.6 milligrams of oxygen per liter day. He points out, however, that the daytime respiration of these organisms consumes one-half of the gross photosynthetic oxygen production. Other worthwhile articles in this general area are ''Primary Production in

Flowing Waters,'' by Odum[37] and the review ''Measuring the Production of Marine Phytoplankton'' by Strickland.[20]

The oxygen production by algae in connection with waste stabilization ponds has interested sanitary engineers for many years. A summary of typical waste stabilization pond experiences was given by Towne, Bartsch, and Davis[38] in 1957. Of even greater interest to those studying impoundment behavior is the work of Hull, however. Hull compares the rate of photosynthetic oxygen production with the surface reaeration in Baltimore Harbor in ''Oxygenation of Baltimore Harbor by Planktonic Algae,''[39] and the Delaware River in ''Photosynthetic Oxygenation of a Polluted Estuary''[40] through the use of the light- and dark-bottle technique. Although this technique is criticized by Pratt and Berkson[41] in ''Two Sources of Error in the Oxygen 'Light' and 'Dark' Bottle Method,'' Verduin[42] in ''Energy Fixation and Utilization by Natural Communities in Western Lake Erie,'' and Kaplovsky[43] in ''Discussion of 'Photosynthetic Oxygenation of a Polluted Estuary','' and the problems with this technique are summarized by Strickland,[20] these studies still give the most direct indication of photosynthetic activity. Hull has used the method, with reservations concerning its accuracy, to show the general significance of photosynthesis in the oxygen balance of polluted waters.

In the Baltimore Harbor work, Hull[39] calculates that in the summertime the algae produce 600,000 pounds of oxygen per day, whereas atmospheric reaeration at the air-water interface provides only 187,000 pounds per day. He also refers to studies of Nusbaum and Miller conducted in San Diego Bay. They calculated the ratio of photosynthesis to atmospheric oxygenation as 1.35/1.

In the Delaware River studies, Hull[40] states that August is the critical month with respect to oxygen resources. The mean photosynthetic oxygen contribution, calculated from 16 measurements during the month, was 1,789,000 pounds per day, which Hull feels is a low estimate. He feels that the maximum rate obtained, 3,893,000 pounds per day, is really more representative of the photosynthetic production on a sunny August day. Since the oxygen demand in the river during the same period was calculated as 3,260,000 pounds per day, the atmospheric reaeration must have been somewhere between 45 and 0 percent of the total demand, depending on the actual photosynthetic contribution.

Hull's general thesis in all of his writings is that photosynthetic oxygen is a source of oxygen that has long been overlooked and/or neglected by sanitary engineers. This concept is expressly contained in a discussion by Hull[44] of a paper by O'Connor on reoxygenation of water.

In some instances photosynthetic activity is so vigorous that the upper layers become supersaturated, and oxygen is lost to the atmosphere. An analysis of Roanoke Rapids Reservoir by Pritchard and Carpenter, included as an appendix in the report by Fish, Hull, Peters, and Knight,[45] shows a loss of 1.0 gram of oxygen per square meter per day to the atmosphere. Hull[46] analyzed the same data by a different technique but with the same result. He concludes:

> "There appears to be a need for study to find ways of capturing the oxygen lost to the atmosphere. Perhaps mechanical mixing of the reservoir water to prevent supersaturation at the surface is one method of retaining and using for waste decomposition the oxygen now lost to the atmosphere. If the average surface oxygen concentration can be reduced by this means to values significantly below saturation, the atmosphere will become an additional oxygen source instead of an oxygen sink."

Attention has also been given in the literature to the oxygen demand of the algal population in water. This demand takes three forms: (1) respiration that occurs while photosynthesis is progressing, (2) respiration that occurs at night when photosynthesis is absent, and (3) the oxygen uptake caused by consumer organisms, mainly bacteria, that metabolize the algal bodies upon their death.

As mentioned above, Verduin[31] suggests that the daytime respiration is about one-half of the gross photosynthetic rate. In the same article he also states that the balance between the net daytime photosynthetic rate and the nighttime respiration of the algal community is fairly close. This means that if all of the algae stayed in the upper waters of an impoundment, the net 24-hour contribution to the oxygen balance would be near zero. Because many algae fall from the euphotic zone during a given 24-hour period, however, the dark respiration, and/or the oxygen uptake because of bacterial decomposition, do not occur in the surface waters. The net increase in oxygen concentration is, therefore, often in the surface waters because of algal action, since the major portion of the oxygen demand takes place in the bottom waters or in the bottom muds.

Several investigators have discussed this problem. Mackenthun[21] quoting Renn, states, ''The nutrients' stimulation of algal production can lead to the formation of a mass of organic matter greater than that of an original waste source.'' Knopp[47] in ''The Influence of Plankton on the Oxygen Demand of River Water'' shows that an error of more than 50 percent in the overall balance might occur if the oxygen demand caused by the decomposition of algae were neglected. Thomas[48] in his article on ''Oxygen Deficit Rates for the Central Basin of Lake Erie'' discusses the measured rate of oxygen depletion in the hypolimnion of central Lake Erie. He found that this rate was 0.037 milligram per square centimeter per day and that this was very close to the maximum possible rate that could be sustained and still maintain dissolved oxygen throughout the hypolimnion during the stagnation period. This demand was partially caused by the decomposition of algae in the lower waters of the lake. Edmondson[49] in his article ''Secondary Production and Decomposition'' also addresses himself to the problem of the growth of consumer organisms in lakes and the oxygen demand caused by these organisms.

Hull's[46] position is summarized in the following quotation:

> ''I have been frequently reminded that for every gram of oxygen released by photosynthesis, a gram of biochemical oxygen demand is produced by the synthesis of organic matter, and therefore there can be no net gain of oxygen. This is a gross oversimplification-it must be qualified by consideration of the relative rates, periods, and locations of the oxygen liberation and subsequent oxygen demand. To support this, one need only point to the vast standing crop of fixed carbon on this earth in the form of petroleum, coal, plants, and animals, including several billions of human beings. All of this carbon was fixed by photosynthesis, but some of it has waited millions of years to be oxidized, before it will eventually take back the oxygen literated at its birth.''

Research Needs

Two quotations from the literature indicate the research needs in this area. Mackenthun[21] states, ''There is a need to relate photosynthetic oxygen production to: (1) practical problems associated with lakes and streams, (2) some quantitative expression of phytoplankton, (3) nutrient balance, and (4) seasonal fluctuations and climatic phenomenon.'' Verduin[50] in ''Photosynthesis by Aquatic Communities in Northwestern Ohio'' remarks, ''Any attempt to correlate metabolic rates under natural conditions with environmental factors (nutrients, temperature, light supply, etc.) yields 'shotgun results'.''

The overall role of algae in the production of oxygen and the oxygen demand of impounded waters, as related to the environment, must be determined if the behavior of impounded waters is to be completely understood. Past studies must be extended in scope to include the production of oxygen in surface waters, the demand for oxygen in lower waters, and the transfer of oxygen at the surface so that the entire oxygen balance of an impounded body of water is truly measured. Although the light- and dark-bottle technique has its disadvantages, this method is an attempt to actually measure *in situ* oxygen concentration changes. Some method must be developed to overcome the difficulty of unnatural conditions that occur in glass bottles, e.g., the lack of mixing. Also, these data must be taken over a short enough period of time so that the environmental conditions, sunlight, water temperature, wind velocity, and so forth, will be steady during the time the measurements are being taken. Only through this technique can the influence of algae on the oxygen balance of impounded water, as related to the environment, ever be determined. This determination must be made, for surely the relationships between algal development and the environment are not as haphazard as would be suggested by Verduin.[50] A study designed to develop equipment to make these measurements with the use of dissolved oxygen probes has begun in the Engineering Section at the Robert A. Taft Sanitary Engineering Center.

CARBONATE EQUILIBRIUM

Existing Information

Since algae remove carbon dioxide from solution as they grow and convert it into cellular material, the carbonate equilibrium as it exists in natural waters is affected.

Some work by Sawyer[50a] shows why the pH attained in natural waters by algae can exceed 7. He has observed values as high as 9.8 where algae are in high concentration and under favorable light conditions. The equilibria involved are shown in Equation 1.

$$(2\ HCO_3^- \rightleftharpoons CO_2 + [CO_3^= + H_2O) \rightleftharpoons HCO_3^- + OH^-] \qquad (1)$$

When carbon dioxide is removed, the primary equilibrium shown in parentheses ( ) shifts to the right and the carbonate ion concentration increases. An increase in carbonate ion concentration, however, effects the secondary equilibrium, shown in brackets [ ]; the secondary equilibrium also shifts to the right and the hydroxide ion concentration increases. The pH, therefore, rises to levels considerably above 7. A ceiling on pH is formed by precipitation of calcium carbonate that controls the amount of carbonate ion that can coexist in solution with calcium ion. See also Weber and Stumm.[50b]

Neel, Nicholson, and Hirsch[8] also discuss this same set of equilibria. They point out that carbonate precipitation is common in standing bodies of water and has been responsible for commercially important marl deposits in natural lakes. They also note that the transformation of bicarbonate to carbonate ions occurs in upper layers of the water and that in a stratified impoundment the precipitated carbonate particles dropping down into the hypolimnion may or may not reach the bottom. This is because the hypolimnion generally contains relatively higher concentrations of carbon dioxide because of the biological activity that has occurred in these waters during the decomposition of falling algal bodies and because of the absence of many viable algae in the hypolimnion. The precipitated carbonate particles combine with this carbon dioxide and are converted into soluble bicarbonates that increase the alkalinity of the deeper waters. With these relationships in mind, these investigators measured, reported, and discussed the changes in alkalinity and in hardness that occurred in the six reservoirs on the Missouri River.

Limnologists have often shown an interest in the carbonate equilibrium in natural waters because of the possibilities of measuring primary production by the change in carbon dioxide concentration in the waters. Verduin[42] is very enthusiastic about this procedure, feeling that this is the only method to measure primary production in unconfined waters. He further states that measurement of photosynthetic rates in unconfined waters is the only method of obtaining the true rate. In his opinion, whenever a sample of water is enclosed in a container, it no longer represents the natural environment, and measurements of the sample, therefore, do not represent natural photosynthesis.

For example, the gross photosynthetic rate of 16 grams of oxygen per square meter per day measured in unconfined water, as cited by Verduin,[31] is much greater than the rate of 2.25 grams per square meter per day that Hull[46] reports in the upper 0.5 foot in the Delaware River Estuary at 20°C. These data are from light- and dark-bottle experiments. Hull states that this value is related to temperature, however, and at 30°C he gives a rate of 21 grams of oxygen per square meter per day.

Research Needs

Research in this area is important, since changes in the carbonate equilibrium can cause either the precipitation of hardness-causing ions, the reduction of alkalinity, or under reverse conditions the dissolution of carbonates from the impoundment bed. Any complete understanding of the behavior of impounded waters should include some knowledge of the influence of microorganisms, both algae and bacteria, on the carbonate equilibrium. Some of the studies mentioned above, designed to relate the growth of an algal population to a specific environment, should also include studies on the carbonate equilibrium in the same environment circumstances.

NUTRIENT REMOVAL

Existing Information

Since algae require nitrogen and phosphorus to build new protoplasm, they take these nutrients from solution during their growth cycle. This is also true of other nutrients, but to a smaller extent; therefore, these will be neglected for the moment. If the algae are removed from the water before they die, decompose, and release the nitrogen and phosphorus back into solution, these troublesome fertilizing elements will not be available for subsequent generations of algae. Since nitrogen is generally present in more abundance than phosphorus, and because of the nitrogen-fixing properties of many algae,[17,22] the possibility of removing phosphorus from solution through algal growth has been given the greatest consideration in the literature. Sawyer[17] suggests that at least 10 micrograms of phosphorus per liter of water is necessary for algal growth. This implies that if algae were used for phosphorus removal, the concentration could not be reduced below this minimum level. Hutchinson,[18] however, states, ''Some organisms, such as *Asterionella*, in natural lake waters are extraordinarily efficient in the utilization of the element phosphorus and can take it up from water containing less than 1 μg/l.'' Lackey,[22] quoting Young, states, ''Young (1955) concludes that, 'there is, in any habitat, no one factor which determines the species density of an area, but always a combination of factors.' '' This means that in any nutrient removal scheme there is the possibility that a combination of factors rather than the concentration of a specific nutrient of interest may limit the population density. This would interfere with a specific nutrient removal scheme.

The combination of factors may or may not be a serious problem. Liebig's Law of the Minimum[51] suggests that the quality of the least abundant nutrient relative to the requirement for that nutrient controls the maximum development of a culture, irrespective of the concentration of any of the other nutrients. This means that a group of organisms will grow to some limit imposed by the nutrient in least relative supply for that group, even though the water might be richer in many other nutrients. Even if this group of organisms were removed, a new group could grow in this water because of different nutrient requirements. Therefore, the original nutrient removal attempt would not necessarily protect the impoundment against all further algal blooms.

Some attention has been given to the removal of nitrogen, phosphorus, or both in waste stabilization ponds. Bogan[22] found that algae were effective in removing phosphorus from sewage. In a high-rate controlled laboratory experiment, however, the extremely rapid rates of phosphate removal were caused more by a rising pH and corresponding precipitation of calcium phosphate than by metabolic uptake. In summary, he states, ''Adsorption and coagulation appear to play a major role where rapid removal or large amounts of phosphate is involved [in the high-rate laboratory experiments]. Metabolic conversion is the principal removal mechanism under the more leisurely conditions prevailing in oxidation ponds and sewage lagoons. In the latter case [oxidation ponds] efficiency of phosphate removal is proportional to detention time, other things being equal.''

Gates and Borchardt,[52] authors of ''Nitrogen and Phosphorus Extraction from Domestic Waste Water Treatment Plant Effluents by Controlled Algal Culture,'' have also studied the removal of nutrients from secondary sewage effluent. They included studies on both nitrogen and phosphorus in their research. They investigated performance as related to four variables: (1) culture depth, (2) light intensity, (3) nutrient concentration, and (4) culture density.

They found mixing and maintaining an algal concentration to nutrient concentration of 4 to 1 beneficial in the removal of nutrients. As this ratio increases because of nutrient utilization, the rate of removal decreases, however, so that the degree of treatment and the time required for treatment are not linearly related. In spite of this useful research the authors realize more study is needed since they close their article with this comment: ''Areas which need considerable attention are: separation techniques for algae; techniques for utilizing waste algae; further temperature, depth, and light relationships; the capability of algae to extract trace elements; and the engineering development of mechanisms for the production of light integration.''

Hicks,[53] in 1958 reviewed the methods available for harvesting algae from water. He investigated centrifugation as well as coagulation with alum and acid. Some studies were also made on the anaerobic digestion of the harvested algae. He concluded, ''At present there appears to be no commercially sound scheme for harvesting the algae.'' Bogan[22] also referred to the problem of harvesting algae. He mentioned screening, settling, centrifuging, and chemical coagulation. In summary he states, ''All have been found wanting in some aspect, generally in terms of efficiency and often in terms of cost. From the standpoint of performance and economy, some type of screening device appears to be the most promising.'' This comment agrees with some of the European experiences using floating microstraining devices, such as those developed by Glenfield and Kennedy, Inc.

Oswald[54] in his article ''The Coming Industry of Controlled Photosynthesis,'' which summarized 10 years of his research in the area of growing algae on sewage states, ''Nevertheless, the fundamental findings of the past 10 years are unmistakable. It is technically feasible to apply controlled photosynthesis to treat wastes, to reclaim water, to render waste organic matter suitable for use as an animal feedstuff, to obtain unprecedented yields of protein per acre of pond surface in either fresh or saline water, and to produce electrical power from sunlight using organic wastes as a medium.'' He goes on to remark, however, that in spite of the present technical feasibility of harvesting algae, more research needs to be done to improve the economics of this step of the process.

Research Needs

The research needs in this area center on three points: (1) determination of how efficient a nutrient removal scheme under lake conditions can be, (2) the most effective way of harvesting the algae, and (3) destruction of harvested algae. Laboratory experiments on the growth of algae and their subsequent nutrient removal in lake waters would supply information pertinent to the first point. The harvesting studies comparing existing equipment, with possible modifications, under field conditions would contribute to the second. Some useful information on the ease of destruction of harvested algae could be obtained on a laboratory or small pilot scale. Laboratory, small pilot plant, and field studies on this topic would indicate whether the growing of algae themselves could be an effective method of removing fertilizing nutrients and preventing further growth. Stimulation of fish life might be an area worth exploring as an algae removal method. Another possibility is removing or covering the upper layer of bottom mud, which is usually rich in settled phosphorus, to prevent this nutrient from being recycled into the euphotic zone.

Inorganic Substances

DISSOLVED OXYGEN

Existing Information

The behavior of the dissolved oxygen concentrations in polluted streams has received considerable attention through the years. This attention started with the classic work of Phelps[55] in 1912 and of Streeter and Phelps[56] in 1925. Their work has been reviewed, and modifications have been suggested by various investigators, such as Fair[57] in 1939; Hull[58] in 1960; O'Connor and Dobbins[59] in 1956; Krenkel and Orlob[60] in 1962; and Churchill, Elmore, and Buckingham[61] in 1962. A useful summary was published in 1958 by the U.S. Public Health Service, ''Oxygen Relationships in Streams,''[62]

Unfortunately, similar active interest has not been shown in the dissolved oxygen relationships in impoundments. The role of algae is discussed earlier in this report; Hull,[39,40] and Winberg and Sivko[63] are some of the investigators who have reported on this subject. As stated previously, however, although algae may produce a net increase in dissolved oxygen concentration in surface waters, their decomposition in deeper waters, with a corresponding oxygen uptake, means that the net gain in the oxygen resources of the impoundment is reduced, as noted by Mackenthun.[21]

Even less attention has been given to the atmospheric reaeration capacity of slowly moving or standing bodies of water. Hull[39] calculated this contribution for one set of circumstances in Baltimore Harbor, and Kittrell[64] in his article ''Effects of Impoundments on Dissolved Oxygen Resources'' states that the atmospheric reaeration rate in impoundments is low, but he gives no numerical values for this rate. He does state, however, that in spite of this low rate, the pounds of oxygen transferred through this mechanism may be substantial because of the long detention time. This article is an excellent review of this situation and contains a valuable bibliography.

The existence of low dissolved oxygen in the bottom waters of stratified impoundments has been reported repeatedly since the classic work of Birge and Juday[65] in 1911. Often, this water occurs in impoundments used for peaking hydroelectric power generation. Periodically, a large quantity of water having a low dissolved oxygen concentration is released from these impoundments through the turbines. The problems caused by these releases are discussed by Ingols in ''Pollutional Effects of Hydraulic Power Generation''[66] and ''Effect of Impoundments Upon Downstream Water Quality, Catawba River, S.C.''[67] He points out, that although this low dissolved oxygen concentration in water reduces the downstream dissolved oxygen concentration, it does not necessarily reduce the number of pounds of oxygen downstream because the latter depends upon both volume and concentration. The oxygen-deficient water may well put a burden on the downstream fish population, as well as on a downstream industry or municipality that depends upon the stream for waste assimilation.

Since the problem of releasing water with a low dissolved oxygen concentration through hydroelectric generators has been recognized for some time, various remedial measures have been proposed. Scott, Wisniewski, Lueck, and Wiley[68] in ''Aeration of Streamflow at Power Turbines'' and Bueltman, Termini, and Kingsbury[69] in ''Power Requirements for Oxygen Transfer with Turbine Aerators'' discuss methods of introducing air into the turbines of the power generators. Although the efficiency of electric power generation would be reduced slightly, these authors feel that adding dissolved oxygen at the turbine is fairly economical.

In another method being tried at two impoundments on the Roanoke River, a submerged weir about 400 feet upstream from the power intake is supposed to block the flow of hypolimnion water to the power intake and permit the flow of top waters to the turbines. This installation is discussed in detail by Fish, Hull, Peters, and Knight[45] and by Ragone and Peters.[10]

Olszewski[70] in ''Experiment on Drawing Off the Hypolimnic Water from a Lake'' reports on an experiment in Lake Kortowo in Poland in which the surface outflow was cut off and the resulting increased flow in the epilimnion was used to force the hypolimnetic water out of the basin.

Some proposals have been made for reaerating the stream below a dam. Another method suggested is mixing of the entire reservoir contents. A detailed discussion on this possibility will be given later. Multilevel outlets are discussed by Kittrell;[64] these are effective, but expensive.

Research Needs

The oxygen budget of standing bodies of water must be reevaluated, first to understand what mechanisms of oxygenation and deoxygenation are involved, and second to understand how the environment affects the overall budget. Hull[71] in his ''Second Progress Report of an Investigation of Low-Flow Augmentation for Stream-Pollution Abatement'' reports on a model that was constructed to study atmospheric reaeration in the laboratory. A useful method of studying the behavior of dissolved oxygen at the mud-water interface was described by Fremling and Evans.[72] As previously mentioned, the Engineering Section at the Robert A. Taft Sanitary Engineering Center has begun a field investigation designed to measure more reliably both atmospheric reaeration and photosynthetic contribution in the upper waters of impoundments.

NITROGEN

Existing Information

A very complete discussion of the nitrogen cycle in lake waters is given in chapter 16 of *A Treatise on Limnology* by Hutchinson,[18] and all of the reactions are summarized in Figure 1. In spite of the thoroughness of this review, many of the data quoted, in Hutchinson's opinion, remain only partially explained. For example, in discussing data taken by Mortimer in England, which showed a sudden change in the nitrogen balance of a given lake, Hutchinson states, ''The most reasonable, though not entirely satisfactory, explanation of the phenomenon apparently indicated by Mortimer's results is that at the time of circulation, when the water is oxygenated throughout the lake, the formation of an oxidized microzone permits adsorption of ammonia from the water, and that later this adsorbed ammonia is nitrified and liberated into the lake.'' Later, in discussing similar data taken by Yoshimura in Japan, he remarks, ''It is conceivable that a compensating fall in nonalbuminoid organic nitrogen occurred, either in solution or in the seston, or that the results are due to denitrification or nitrogen fixation. Neither hypothesis, however, seems very likely.''

Hutchinson makes this comment on nitrogen fixation, ''It has become apparent that certain blue-green algae are important nitrogen-fixing organisms, and in the case of genus *Anabaena* there is good reason for believing that such algal fixation is of significance in those lakes in which the genus occurs abundantly.'' He also lists other nitrogen-fixing algae.

One of the investigators who reported in ''Algae and Metropolitan Wastes,'' Dugdale,[22] along with his coworkers, has been studying intensively the phenomenon of nitrogen fixation in lake waters for many years. Authors

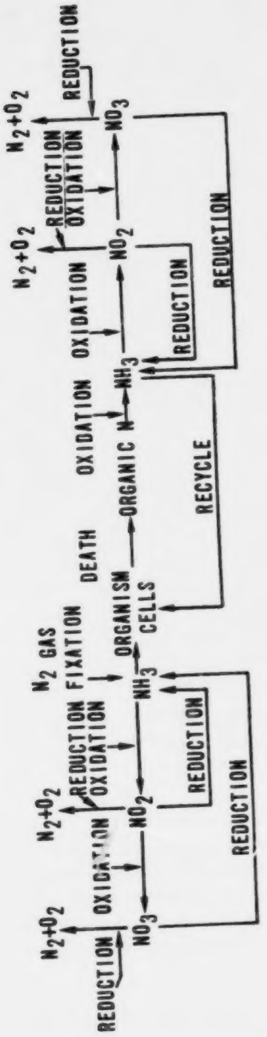

REDUCTION
N_2+O_2
REDUCTION
NO_3
OXIDATION
N_2+O_2
REDUCTION
OXIDATION
NO_2
REDUCTION
REDUCTION
N_2 GAS
FIXATION
NH_3
ORGANISM CELLS
DEATH
ORGANIC N
RECYCLE
OXIDATION
NH_3
OXIDATION
REDUCTION
N_2+O_2
NO_2
OXIDATION
REDUCTION
REDUCTION
N_2+O_2
NO_3
REDUCTION

Figure 1. Summary of nitrogen cycle, after Hutchinson.

in numerous publications feel that nitrogen fixation is a very common and widespread phenomenon in lake waters.

In his summary Hutchinson states that, ''Ammonia is produced by practically all heterotrophic bacteria in the course of organic decomposition. At least in some lakes, nitrification can take place in the free water whenever oxygen is present, whereas nitrate reduction occurs at all times. Nitrification appears to proceed most rapidly in winter and nitrate reduction in summer. The existence of denitrifying bacteria that liberate molecular nitrogen is known, but it seems probable that most of the nitrate reduced forms ammonia.'' Several investigators feel that denitrification is of more importance than Hutchinson implies. Schmidt and Kampf[73] in studies in surface waters, Johnson and Schroepfer[74] in activated sludge studies, and Cooper and Smith[75] in soil studies have all demonstrated significant losses of nitrogen gas during nitrate reduction. There can be no doubt that nitrate is reduced to ammonia when the nitrogen is required for new cell synthesis; but when nitrate acts as an electron accepter during oxidation, the end product is gaseous nitrogen. Finally, to complete Hutchinson's summary, he states, ''Nitrite and hydroxylamine are known to occur in lakes, as intermediates in the oxidation or reduction processes."

Most of the reports on which Hutchinson based his writings were not concerned with the nitrogen balances - probably because nitrogen balances are so difficult to make. He does quote one study by Rohlich and Lea in 1949 in which they calculated that 155.8 metric tons of total nitrogen came into Lake Mendota during a 12-month period, and only 41.4 metric tons of total nitrogen left the lake; thus 114.4 metric tons of nitrogen was not accounted for. This could be caused by denitrification or the settling into the mud of organic nitrogen. If the latter were true, this nitrogen would be available for recycling.

A recent paper by Leclerc,[76] ''Self-Purification of Fresh Water Streams as Affected by Temperature and by the Control of Oxygen, Nitrogen and Other Substances,'' also makes note of the inability to make nitrogen balances in surface waters. He states that rivers receiving normal quantities of sewage self-purify rapidly, but the nitrogen balance in them is incomplete. Later he says that if the total quantity of nitrogen contained in the organic materials discharged to streams were to yield ammonia, and if all this ammonia were to be transformed into nitrite and nitrate, these forms would be much more abundant in streams. He then goes on to suggest a rather novel mechanism for loss of nitrogen gas. He suggests the following reaction between nitrites and amino acids:

$$R\text{-}NH_2 + HNO_2 \rightarrow ROH + N_2 + H_2O \qquad (2)$$

The more common explanation for the lack of a nitrogen balance is the reduction of nitrate to yield nitrogen gas and oxygen, although the uncertainty in the nitrate-nitrogen determination by the phenoldisulfonic acid method may well be the explanation for many of the poor nitrogen balances obtained through the years. In addition to the work in rivers, extensive work on the biological transformations of nitrogen in the activated sludge process has been done by Symons and his co-workers.[77,78,79,80]

Research Needs

The research needs in this area fall into four categories: (1) analysis, (2) measurement of cycle balances, (3) measurement of the effect of the environment on the cycle balances, and (4) rate and extent of nitrogen transformations in a given impoundment. Many analytic problems must be solved before the researcher is confident that he can measure all forms of the nitrogen occurring during the study of these cycles. True nitrogen balances must be measured, for only through the measurement of a nitrogen balance can the investigator be assured that all of the nitrogen has been accounted for at any given time. Once a nitrogen balance can be made accurately, the influence of the environment on this cycle must be investigated. Finally, the effect of the biological transformations of nitrogen must be measured in terms of the other quality characteristics in impounded waters.

For example, little attention has been given to the oxygen demand that the nitrification of ammonia creates. Since the ultimate BOD of ammonia is 4.57 milligrams per milligram, nitrification could severely tax the oxygen resources of an impoundment or stream. The article by Courchaine[81] entitled ''The Significance of Nitrification in Stream Analysis - Effects on the Oxygen Balance'' showed that in the particular instance under investigation, nitrification accounted for more than one-half of the total oxygen demand. Laboratory studies have been started in the Engineering Section at Robert A. Taft Sanitary Engineering Center on all phases of the nitrogen cycle.

PHOSPHORUS

Existing Information

Much less controversy revolves around the phosphorus cycle, possibly because it has been studied so much less than the nitrogen cycle. Chapter 12 of Hutchinson's text[18] stands as a prime authority on this cycle. Based on some radiophosphorus experiments within Linsley Pond in 1950, Hutchinson summarized the phosphorus cycle as follows:

> ''The phosphorus cycle exhibited by these experiments during extreme summer stratification may therefore be considered to involve the following processes.
>
> 1. Liberation of phosphorus into the epilimnion from the littoral, largely from decay of littoral vegetation.
> 2. Uptake of phosphorus from water by littoral vegetation.
> 3. Uptake of the liberated phosphorus by phytoplankton.
> 4. Loss of phosphorus as a soluble compound, less assimilable than ionic phosphate, from the phytoplankton, probably followed by slow regeneration of ionic phosphate.
> 5. Sedimentation of phytoplankton and other phosphorus-containing seston, perhaps largely faecal pellets, into the hypolimnion.
> 6. Liberation of phosphorus from the sedimenting seston in the hypolimnion, or when it arrives at the mud-water interface.
> 7. Diffusion of phosphorus from the sediment into the water at those depths at which the superficial layer of the mud lacks an oxidized microzone.''

Research Needs

Although the reactions involving phosphorus are known, the behavior of phosphorus in a specific impoundment is not predictable. More data from various environments are needed to permit generalization. Analytic techniques, including pretreatment methods, need to be standardized so that all investigators will be reporting the same species of phosphorus. Six species of phosphorus may be considered: total and soluble orthophosphorus, total and soluble polyphosphorus, and total and soluble organic phosphorus.

SULFUR

Existing Information

The sulfur cycle in lake waters is also uncontroversial, chapter 13 of Hutchinson's text[18] being a very useful summary of the subject. In this chapter he notes that under anaerobic conditions sulfate can be used as a hydrogen acceptor in the metabolic oxidation of organic matter by certain bacteria. Hydrogen sulfide is also produced in the bacterial decay of any organic matter, such as protein, that contains reduced sulfur. This reaction often occurs in the deoxygenated hypolimnion of lakes. Since ferrous sulfide is very insoluble in neutral or alkaline water, however, hydrogen sulfide appears in the free water of the hypolimnion only when the water is acid, or when all of the iron has already been precipitated from solution.

Hutchinson goes on to say that in aerated waters, the bottom muds lose sulfur to the water as sulfate. Therefore, in unstratified waters the sulfate concentration increases with depth. In stratified waters, however, the sulfate concentration decreases in the hypolimnion waters, although a sufficient H_2S concentration may not appear in the free water to account for this decline. The latter phenomenon is due to sulfide precipitation as noted above. He states that possible intermediate products of sulfate reduction, such as thiosulfate, seem to be absent in lake waters. To complete the sulfur cycle, Hutchinson remarks that hydrogen sulfide can be oxidized by sulfur bacteria, which use the oxidation as a source of energy. Hydrogen sulfide can also be oxidized by photosynthetic colored bacteria, which use the gas as an oxygen acceptor in the photosynthetic reaction.

Hutchinson closes his summary of the sulfur cycle by stating that the sulfur budget of a lake may be strongly affected by the inflow of highly mineralized water or sulfur-containing industrial wastes. In the presence of these high inputs of sulfur-containing material, the hypolimnion of a lake can often stimulate the generation of tremendous quantities of hydrogen sulfide. A concentration of hydrogen sulfide of 786 milligrams per liter of water at the bottom of Big Soda Lake, Nevada, has been found under the conditions described previously.

Research Needs

As with nitrogen and phosphorus, analytic problems need to be solved to permit the measurement of all forms of sulfur in low concentrations. Once a sulfur balance can be made, the influence of the environment on it must be measured to allow future predictions. Knowledge of hydrogen-sulfide-producing conditions are important because hydrogen sulfide in high concentrations may be toxic to the organisms and the discharge of waters laden with hydrogen sulfide through power dams will cause problems.

IRON

Existing Information

Not too much concerning the iron cycle in impounded waters appears in the literature. Hutchinson[18] treats this matter in chapter 11 of his book, as do Stumm and Lee[82] in their article ''The Chemistry of Aqueous Iron.'' In a section of their paper entitled ''Some Limnological Aspects,'' Stumm and Lee summarize the iron situation in impounded waters. This summary, which draws heavily on Hutchinson, is pertinent to this report. They state:

> ''A vast amount of analytical data exists relating to the iron content of inland waters. The uncertainty of analytical techniques used in these investigations is a serious obstacle for the interpretation of the available data. The published analytical techniques do not permit a rigorous quantitative differentiation between ferrous and ferric iron. Furthermore, it is rather difficult to distinguish analytically dissolved and suspended iron.
>
> ''Bearing in mind the theoretical conclusion of the solubility equilibria, the available data indicate that a substantial fraction of the iron in many lake waters is present either in suspended form or as an organic iron complex. However, the waters of uncolored neutral or alkaline lakes, which are poor in plankton, have iron contents usually ranging from 50 to 200

> μg/l, which appears to be in accord with solubility equilibria. In the epilimnion of such lakes where generally the concentration of dissolved oxygen is high, the concentration of soluble ferric iron cannot exceed 18 μg/l and ferrous iron is unstable in the presence of dissolved oxygen. Any iron which enters the epilimnion will eventually be precipitated as ferric hydroxide. The insoluble ferric iron settles into the hypolimnion.
>
> ''Here, in the absence of dissolved oxygen, favorable conditions for the reduction of ferric hydroxide to ferrous iron may prevail. In waters free of oxygen, ferrous iron may be soluble up to a few mg/l. Except on occasion at its surface, the mud at the bottom of lakes probably always has reducing properties and iron which has not been reduced in the water layers overlying the mud is reduced to the ferrous state. This ferrous iron will diffuse through the mud or be displaced by consolidation of the deposits and will eventually reach the aerobic zone.
>
> ''Upward transport may be promoted by the formation of methane or hydrogen sulfide in that portion of the deposits that is undergoing anaerobic decomposition. During stagnation periods, the iron content of the water overlaying the deposits will progressively increase. Ferrous iron which is transported through upward diffusion into the oxygenated waters near the surface will be oxidized to ferric hydroxide.''

Figure 2 summarizes their thinking.

Although microorganisms are responsible for the absence of oxygen in bottom waters and bottom deposits, the iron reactions mentioned by Stumm and Lee are all purely chemical. A paper by Morgan and Stumm,[83] ''The Role of Multivalent Oxides in Limnological Transformations as Exemplified by Iron and Manganese,'' reiterates this cycle and adds important new information on the rates of oxidation of iron under differing environmental conditions.

Research Needs

Although some attention has been given to the iron cycle in impounded waters, further research is needed to delin-

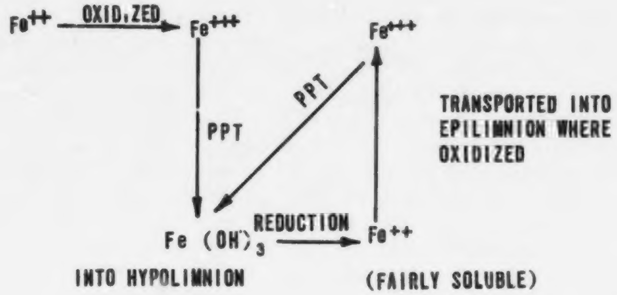


Figure 2. Summary of iron cycle, after Stumm and Lee.

eate the effect of the environmental conditions in the overlaying water and in the bottom mud as they relate to the rate and extent of the dissolution of iron. Such factors as soil type, soil iron content, pH, alkalinity, chemical constituents of the overlaying water, oxidation-reduction potential, and bacterial content all undoubtedly have an influence on this dissolution. This influence must be evaluated.

Laboratory studies have been started by the Engineering Section at the Robert A. Taft Sanitary Engineering Center with the object of showing the influence of various environmental factors on the dissolution of iron. Initial studies will be made in soil-water contact columns quite similar to those described by Sylvester and Carlson[84] in their report on the Howard A. Hanson Dam on the Green River in Washington, which was prepared for the Corps of Engineers.

MANGANESE

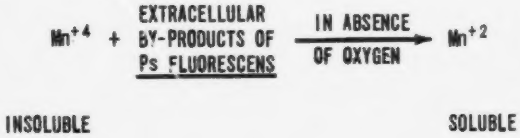
Existing Information

The manganese cycle in impoundments seems to have received much more attention in the literature than iron, possibly because of the greater difficulty in removing it during water treatment. Nine references starting in 1928 and proceeding to 1941 were concerned with manganese concentrations in and removal from bottom waters of deep impoundments. More recent work, however, by Ingols and his co-workers have sought to investigate the mechanism by which manganese is dissolved in impounded waters.

Recent attention has been given to the problem of high manganese concentrations in bottom waters by Churchill,[6] Love,[7] and Ingols.[67] Flentje[85] in an article entitled ''How is Your Manganese?'' states that manganese in storage impoundments is present in more areas than is suspected. He points out that impoundments in only nine states are probably immune from manganese and that approximately 0.1 percent of the earth's crust is manganese. The problems caused by manganese are reviewed by Myers[86] in ''Manganese Deposits in Western Reservoirs and Distribution Systems.'' He notes that a reservoir may show a continual manganese concentration of less than 0.1 milligram per liter of water, yet the manganese can cause discoloration and lead to deposits in pipe lines and filter plants. He also remarks that the reaction of manganese and orthotolidine to give a color, and therefore a false chlorine residual, may seriously interfere with maintenance of safe water throughout a distribution system.

In their review ''Mechanism of Manganese Solution in Quiescent Lake Waters,'' Ingols and Wilroy[87] suggest a mechanism of manganese solution in which trivalent manganese intracellularly accepts electrons and is reduced to soluble divalent manganese, which diffuses out of the cell. Ingols' current (1964) thinking, as reported in a private communication, is somewhat different, however.

Experimentation with *Pseudomonas fluorescens* showed that dissolution occurred when cell-free filtrates of a culture growing in the absence of manganese dioxide were added to a medium containing manganese dioxide crystals. All oxygen was removed from this medium by an actively growing *E. coli* culture, which was incapable of dissolving manganese. After 48 hours, detectable quantities of soluble manganese indicated that the extracellular by-products of the *Ps. fluorescens* could reduce manganese dioxide. Figure 3 summarizes the dissolution.



$$\underset{\text{INSOLUBLE}}{Mn^{+4}} + \underline{\begin{array}{c}\text{EXTRACELLULAR}\\ \text{BY-PRODUCTS OF}\\ \text{Ps FLUORESCENS}\end{array}} \xrightarrow[\text{OF OXYGEN}]{\text{IN ABSENCE}} \underset{\text{SOLUBLE}}{Mn^{+2}}$$

Figure 3. Manganese dissolution, after Ingols.

The concepts of Morgan and Stumm[83] are shown in Figure 4. Their data indicate that a considerable quantity of manganous ion adsorbed to manganic or ferric oxide particles is carried into the hypolimnion. This means that all of the manganese is not oxidized, even in the upper water where dissolved oxygen is present. Manganous ion is released into solution in the hypolimnion as a result of anaerobic bacterial activity in these waters. Morgan and Stumm agree in general with Ingols et al. that bacterial activity is involved in the manganese cycle.

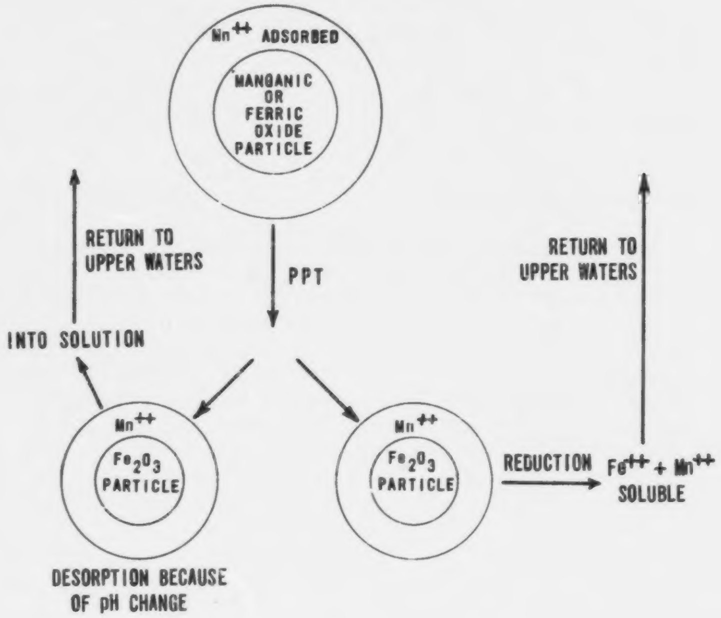

Mn^{++} ADSORBED
MANGANIC OR FERRIC OXIDE PARTICLE
RETURN TO UPPER WATERS
PPT
RETURN TO UPPER WATERS
INTO SOLUTION
Mn^{++}
Fe_2O_3 PARTICLE
Mn^{++}
Fe_2O_3 PARTICLE
REDUCTION
$Fe^{++} + Mn^{++}$
SOLUBLE
DESORPTION BECAUSE OF pH CHANGE

Figure 4. Summary of manganese cycle, after Morgan and Stumm.

Research Needs

Research needs related to manganese dissolution are similar to those mentioned in the iron section. Some additional study is needed to delineate the environmental factors that control the rate and extent of this dissolution.

Some attention to environment was given by Wilroy and Ingols[88] in ''Aging of Waters in Reservoirs of the Piedmont Plateau'' in which they studied three impoundments in a single location near Newnan, Georgia. At the conclusion of this study these authors expressed the opinion that, ''the manganese content of the bottom waters is a function of soil type.''

Laboratory studies have been started by the Engineering Section at the Robert A. Taft Sanitary Engineering Center with the object of showing the influence of various environmental factors on the dissolution of manganese. Initial studies will be made in soil-water contact columns quite similar to those described by Sylvester and Carlson.[84]

NUTRIENTS

Existing Information

Any discussion of pollutional characteristics must recognize the pollutional potential of nutrient materials. Although nutrients are generally classed as pollutants because they support nuisance biological populations, sometimes they make water unfit for a specific use merely because of their presence. Many organisms, for example, slime-forming bacteria, aquatic weeds, rough-fish, and certain insects and insect larva can form nuisances, but the primary nuisance organisms stimulated by nutrient pollution are algae. The problem of increased algal productivity because of nutrient pollution was covered previously, and is not repeated in detail here. One excellent review of this subject, however, is ''Algae and Metropolitan Wastes,''[22] the transactions of a seminar held on April 27-29, 1960, at the Robert A. Taft Sanitary Engineering Center.

In Committee Print No. 29 of the Select Committee on National Water Resources, United States Senate, which was prepared by Reid,[3] and titled ''Methods of Approximating Dilution Water Requirements as a Supplemental Measure for Control of Water Quality in Rivers,'' three pollutional problems are discussed. The first relates to oxygen-demanding materials; the second deals with pollution from plant nutrients; and the third concerns pollution due to persistent chemicals. To quote from the conclusions, ''Three techniques for dealing with different kinds of water pollutants have been described. The limiting one in each instance should be the key to limits on future river developments. In general, the first, oxygen degradation, will become increasingly less important, as the degree of waste treatment increases, giving way to the second and perhaps eventually to the third.'' Therefore, Reid suggests that in the near future the major problem in river pollution will be caused by pollution from plant nutrients.

Research Needs

The research needs in this area have been discussed and are repeated here only in summary form. The prime research need in this area is to develop a fundamental understanding of the true nutrient requirement of some of the more common species of algae so that the relationship between eutrophication and the development of algae will be more clearly understood, or to develop a reliable operational test to determine the algal growth potential (AGP) of a given water.

Organic Substances

OXIDATION AND PERSISTENCE OF ORGANICS

Existing Information

The organic content of impounded bodies of water has long been recognized. Hutchinson[18] in chapter 17 of his text and Derby[89] in his article ''Chlorination of Deep Reservoirs for Taste and Odor Control'' are two among several authors who have written about the subject. Derby's article notes that during thermal stratification of impoundments the creation of anaerobic bottom waters forms an environment in which taste- and odor-producing materials can result from decomposition of dead algae, weeds, leaves, and fish.

In spite of this attention to the organic content of impoundments, very little has been written specifically on the subject of the oxidation of organics in impounded waters. Hull[90] in his ''Bibliography on Biochemical Oxygen Demand'' has gathered together 314 references on this subject, covering the years 1927 through 1961. Although several of the articles were about oxidation of organic matter in the sea, the bulk of the articles concerned the BOD in flowing streams, only four references out of the 314 being directly applicable to impoundments. Ridenour,[91] who wrote "Studies on the Effect of a Small Impounding Reservoir on Stream Purification" found that in a small pond with a hydraulic detention time of about 3 days, calculations yielded a $k_1 = 0.168$ for deoxygenation and $k_2 = 0.26$ for reaeration. He felt that this indicated a more intense rate of deoxygenation and about the same rate of reoxygenation as could be expected in a flowing river.

Mohler[92] in his article "Assessment of the Self-Purifying Power of Surface Waters" discusses the results of observations over a 5-year period on the Aare River. He states that self purification has been improved by reduction of the speed of flow and the increase in the surface area. He concludes by saying, "Creation of impoundments does not necessarily have an adverse effect on self-purification." Zobell and Stadler[93] in "The Effect of Oxygen and Tension on the Oxygen Uptake of Lake Bacteria" show that although the oxygen tension influences the multiplication of lake bacteria, their respiration is independent of the oxygen tension within the examined ranges of 0.3 to 36.5 milligrams per liter. Other useful articles in this general area are "Organic Pollution of New York Harbor - Theoretical Considerations" by O'Connor,[94] "Mechanism of Self-Purification in Flowing Waters" by Jaag,[95] and "Investigations on the Influence of Retention by Dams on Natural Self-Purification" by Liebmann.[96]

No direct consideration was found in the literature of the persistence of organic matter in impounded waters, but a potential problem was indicated in a paper entitled "Biodegradation of Surfactants Under Aerobic and Anaerobic Conditions" by Wayman, Robertson, and Hall.[97] In this article the authors note that under anaerobic conditions, synthetic detergents that are commonly accepted to be biodegradable aerobically were not decomposed. This same effect was noticed by Robeck and his co-workers who reported that ABS was not metabolized in water-saturated soils,[98] but was metabolized in water-unsaturated soil columns.[99] Since anaerobic conditions may be present in the bottom of a deep, thermally stratified impoundment, synthetic organic chemicals, which may not cause a problem in aerobic waters because of their ability to be biodegraded, may persist in anaerobic bottom waters. A discharge of these waters, through power dams for example, might release a slug of this pollution to the downstream area.

Some organics are persistent or slowly degraded even in aerobic surface water, and the possible toxicity of these organics to aquatic life must not be overlooked. Extensive work on the toxicity of refinery effluents to fish life has been carried out by Irwin and his co-workers. An example of his work is contained in the article ''Evaluation and Relative Resistance of Sixteen Species of Fish as Test Animals in Toxicity Bioassays of Petroleum Refinery Effluents'' by Douglas and Irwin.[100] A valuable bibliography also appears with this article.

Research Needs

The rate of oxidation of organic matter in impounded waters, and the possibility of the persistence of organic materials that are ordinarily biologically decomposable must be further evaluated. Organic pollution is likely to exist in most impoundments, and its behavior must be understood if the overall behavior of the impoundment with respect to water quality is to be known. Some laboratory and field work with such devices as the new total organic carbon (TOC) analyzer developed by the Dow Chemical Co. may help understand the true rates of oxidation of organic matter in impounded waters.

A 500-gallon, 3-foot-diameter, 10-foot-deep pilot plant has been developed by the Engineering Section at the Robert A. Taft Sanitary Engineering Center for the artificial creation of thermal stratification in a water column. With this apparatus, laboratory studies have been started on the rates of oxidation or persistence of artificially induced contaminants, or both, in thermally stratified water. This laboratory apparatus, as well as the previously mentioned field studies, should provide useful information in an area where further research is needed.

METHANE

Existing Information

The formation of methane in the bottom muds of marshes and swamps has been recognized for many years. In spite of this, however, very little attention has been given in the literature to evaluating the true meaning of this mechanism in the overall carbon cycle in standing bodies of water. This is unfortunate, since methane fermentation is an important mechanism by which carbon is truly lost from an impoundment environment.

Algae give off oxygen while converting carbon dioxide, an inorganic carbon source, into new cellular protoplasm which is organic carbon. Bacteria use the evolved oxygen to metabolize organic matter, and in so doing produce carbon dioxide. Therefore, bacteria change organic carbon into inorganic carbon. If no oxygen is added from an outside source, and no carbon dioxide is lost to the atmosphere, this cycle is unbreakable, and no true removal of carbon occurs, although as Hull[46] points out, the the relative rates of these reactions are important. If, however, the algae or the bacteria in the above mentioned cycles settle into the bottom muds and undergo methane fermentation, and the methane escapes, carbon will truly be lost from the cycle.

One recent attempt to evaluate methane production in waste stabilization ponds was reported by Oswald, Golueke, Cooper, Gee, and Bronson[101] in their article, ''Water Reclamation, Algal Production, and Methane Production in Waste Ponds.'' The rate of methane production was measured in the bottom muds of 4- and 7-foot-deep waste stabilization ponds. Under optimum conditions, 8 to 10 cubic feet of gas per pound of volatile solids applied could be obtained; this figure is roughly equivalent to the gas production for anaerobic digestion of sewage sludge. The gas contained a high percentage of methane, but, surprisingly, contained very little carbon dioxide. The main problem discovered was one of temperature. Methane fermentation is very temperature-sensitive; and in ponds in which the temperature did not exceed 14°C, the rate of methane fermentation was low. Since sewage sludge digesters are commonly heated to 35°C to obtain rapid rates of methane production, methane fermentation would be expected to be slow in cold water. This dependence on temperature is particularly pertinent to the conditions in stratified impoundments.

Research Needs

The research mentioned above was done in oxidation ponds, and therefore may not be strictly applicable to deep impoundments. Measurements of the rate and extent of methane fermentation must be made in the hypolimnion waters of deep impoundments. As mentioned by Oswald, et al.,[101] under low-temperature conditions, the rate of methane fermentation is slow. Because of the extremely long water detention time in large impoundments, this slow rate may still allow for a significant quantity of methane production. In a large impoundment, methane fermentation could be an important mechanism in the carbon cycle, even in cooler waters. The validity of this statement must be checked with further laboratory and field investigations on methane fermentation.

Currently (1964) Brezonik is carrying on Doctoral Research under the direction of Lee, at the University of Wisconsin, on the concentration and distribution of oxygen, nitrogen, methane, and carbon dioxide in lake waters. These gases will be trapped and analyzed by gas chromatographic techniques. Results of this study should be quite valuable.

COLOR

Existing Information

Major work is currently (1964) being done on the nature of natural color by Shapiro at The Johns Hopkins University and by Black at the University of Florida. The work by Shapiro seems the more pertinent for the purposes of this paper. In 1960 Shapiro[102] prepared a project proposal that led directly to the Research and Training Grants Branch Project "Inorganic - Organic Interactions in Natural Waters," Grant Number WP-00199. This project started in 1961 and is continuing. The following quotation from Shapiro's unpublished project proposal is included with the author's permission:

> "The importance of the dissolved organic matter as a part of the aquatic environment is becoming more and more apparent. Not only is the dissolved organic matter quantitatively important (being up to 20 times the suspended organic matter, including organisms), but it must play a significant role in the environment, for limnologists and oceanographers are realizing that a knowledge of only the inorganic constituents of water cannot fully explain the phenomena observed. Therefore, it is essential that this hitherto neglected factor be given some attention.
>
> "One fraction of the 'dissolved organic matter, which, because it is so readily apparent, has long generated speculation, is the yellow color-

ing matter of 'humic' matter which is found in all natural (fresh) water.

''Work by the principal investigator over the past nine years has led to the elaboration of techniques for the safe and large-scale recovery of these substances and has elucidated many aspects of their nature, role and distribution. It is proposed that this work be continued in hope of ultimately providing answers to the many unanswered questions of fresh water biology and chemistry.

''*Specific Aims*

''The following aspects will be included in the study which will be restricted almost exclusively to fresh water.

''1. Physical state of the coloring matter.

''2. Chemical nature of the coloring matter.

''3. Relationships between color and the inorganic constituents of the water.

''4. Significance of the above, especially Number 3, to the plankton organisms.

''5. Comparison of waters from different parts of the world.''

Research Needs

It is difficult to evaluate the research needs in this area because the important study by Shapiro has not been completed. His first publication, ''Effect of Yellow Organic Acids on Iron and Other Metals in Water,''[103] discusses the ability of color to peptize iron and hold it ''in solution'' in concentrations much above the limit based on solubility. This work relates to Specific Aim Number 3 above. Although this is only the first publication, it is significant to note that among his five specific aims, no mention was made of investigating the rate and quantity of color that might be leached from the bottom muds in a deep-water impoundment. Therefore, in spite of Shapiro's contribution to the nature of freshwater color and its limnological significance, the scope of his work apparently leaves at least one area of importance still uncovered.

Physical Factors and Environmental Influences

THERMAL STRATIFICATION

Existing Information

Thermal stratification of impounded bodies of water has received a great deal of attention in the literature. Hutchinson[18] in chapter 7, ''The Thermal Properties of Lakes,'' treats the subject exhaustively. One major interest in thermal stratification, which is a layering of water based on temperature-induced density differences, has been the accompanying chemical stratification. Some interest has been shown in the use of cool bottom waters in a stratified impoundment for cooling purposes, but most attention has been given to the influence that thermal stratification has on the chemistry and microbiology of an impoundment.

Sometimes chemical stratification can occur even when thermal stratification does not occur. For example, when a highly mineralized, therefore denser, influent flows along the bottom of a large impoundment, chemical stratification takes place. Such a case is the exception rather than the rule, however.

The classical pattern of summer thermal stratification in the northern half of the United States is reviewed clearly by Kittrell[104] in his paper ''Thermal Stratification in Reservoirs.'' In his summary he indicates the following: in the late winter or early spring the impoundment water is cold, is easily mixed by wind action, and has a uniform temperature from top to bottom. As the season advances and the atmospheric temperature becomes higher, both the inflowing tributary water and the surface water in the impoundment get warmer and warmer, and become more resistant to mixing. Finally, the resistance to mixing becomes great enough to overbalance the ability of the wind to accomplish circulation to the bottom of the impoundment, and stable stratification is established.

The epilimnion may be 5 to 50 feet deep depending on impoundment depth and other factors. The impoundment is mixed to this depth by wind-induced currents, and this layer has a uniform temperature. Below the epilimnion is the thermocline, which may be 5 to 20 feet deep. The thermocline is a stratum in which the temperature decreases rapidly as depth increases. A thermocline has been defined for convenience, though on a strictly arbitrary basis, as a stratum of water in which the temperature decreases 1°C or more in the depth of 1 meter, or 0.55°F per foot. The decrease in temperature is accompanied by an increase in density, with a corresponding increase in resistance to mixing. In the hypolimnion the water is protected from the atmosphere by the overlying thermocline and epilimnion; therefore, very little increase in temperature occurs after stratification is established. Temperatures are low and quite uniform throughout the hypolimnion.

Summer stratification persists until the fall when the influent water becomes cooler. The cooler water is mixed throughout the epilimnion and the upper portions of the thermocline by convection and wind action. Cooling continues until the temperatures and densities of the epilimnion and thermocline approach those of the hypolimnion. When this is accomplished, resistance to mixing is diminished, and the fall overturn occurs.

In northern climates, winter stratification can occur because 0°C water is lighter than 4°C water. This density difference can produce thermal stratification, although now the colder water is the top water. The prevention of wind mixing because of ice cover maintains this stratification, if such a cover exists.

As reported by Kittrell,[104] the difference in density per °C is much greater at higher temperatures than it is at lower temperatures. As shown in Figure 5, the difference in density between 25° and 30°C water (1.40 mg/ml) is 2.34 times larger than the difference in density between 10° and 15°C water (0.6 mg/ml). This means that even if an impoundment showed a rather modest temperature gradient that never reaches the intensity of a thermocline, it could still be density stratified. Generally warmer waters stratify with a smaller temperature difference than colder waters. This factor is specifically discussed by Marlier[105] in his article ''The Biology of Tropical Lakes,'' in which he notes extreme stability in lakes having a water temperature near 25°C.

Research Needs

Although the thermal cause and the chemical effect of thermal stratification have been adequately studied and reported, the effect of physical factors and the environment on the presence or absence of thermal stratification has received little attention.

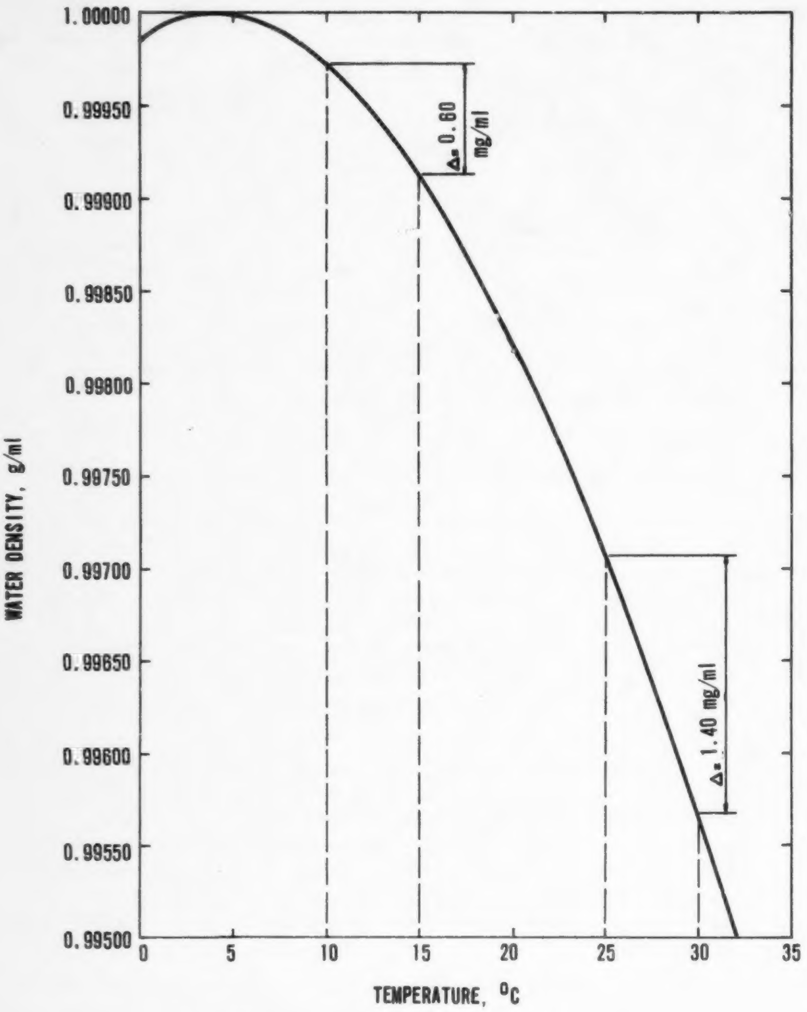


Figure 5. Water densities for various temperatures.

Previous research has not enumerated which environmental factors tend to enhance or inhibit thermal stratification. This situation must be remedied if the site of a planned impoundment is to be evaluated on the basis of its thermal stratification potential. In studies under way in the

Engineering Section at the Robert A. Taft Sanitary Engineering Center several environmental factors are to be correlated with the thermal situation in existing impoundments. Some of the factors to be considered are: (1) general soil characteristics, (2) annual rainfall, (3) average precipitation from April through September, (4) prevailing wind directions, (5) average wind velocity, (6) land cover of surrounding area, (7) time of last frost in spring, (8) time of earliest frost in fall, (9) shape, (10) size, (11) depth, (12) orientation of impoundment, (13) inflow pattern, and (14) weather during the period of the onset of stratification. The correlation of these factors through the use of a discriminate function, to the presence or absence of thermal stratification in existing impoundments will reveal which factors have the major influence on thermal stratification.

DENSITY FLOWS

Existing Information

Density flows, which are defined as streams of water that flow under, over, or through water of slightly different density, have long been recognized in stratified impoundments. The earliest detailed reference found was Wiebe[106] who discussed these in 1939 in his paper entitled "Density Currents in Norris Reservoir." He refers, however, to work done by Monish and Keulegan in 1930, and so density currents were being studied at least that long ago.

In a thermally stratified impoundment with light water on the top, heavy water on the bottom, and intermediate density water in the thermocline area, an influent of a given temperature either flows across the bottom if it is cold, across the top if it is warm, or through some intermediate zone if it is intermediate in temperature. These flows are named underflow, overflow, or interflow, respectively.

Density flows have been used to prevent siltation by discharging silt-laden water through an opening at some specified level in a dam face. The power industry has also made use of density flows by choosing an appropriate level from which to draw cooling water, thereby maintaining the flow of a certain temperature water into their steam-generating plants.

The density currents in Lake Mead, caused by sediment load, not temperature differences, have received attention because of the siltation of the lake. An extensive survey of Lake Mead was begun in January 1948 and completed in late 1950. This was carried out jointly by the Geological Survey, the Bureau of Reclamation, and the Bureau of Ships of the Department of the Navy. The three-volume report published jointly in 1954[107] and the condensed version published by the Geological Survey[108] are worthy of review, not only because of the interesting analysis of the density currents, but also because of the intense discussion of the physical, chemical, and biological changes caused by the formation of Lake Mead.

The Task Committee on Preparation of a Sedimentation Manual of the Committee on Sedimentation of the Hydraulics Division of the American Society of Civil Engineers is publishing portions of the proposed manual in the *Journal of the Hydraulics Division*. The portion ''L. Density Currents''[109] is a very useful review of the subject and contains a good bibliography.

Churchill of TVA reported in a private communication that in the fall season, cooler inflows to Cherokee Reservoir passed through the power intakes in about 2.5 weeks when the theoretical hydraulic detention time of the reservoir, V/Q, was much longer. This type of underflow, or interflow, can be quite striking. A turbid influent can be seen flowing a certain distance into an impounded body of water and then disappearing. The plunge point - where the water, seeking a zone with the same density, dips down into the bulk of the liquid - is often so sharp that a fairly straight line of floating debris accumulates across the surface of the reservoir along the line at which the influent water begins to submerge.

Research Needs

One current piece of research on this problem is being carried on at the California Institute of Technology with support of the Research and Training Grants Branch of the U.S. Public Health Service's Project WP-00248. The title of this project is ''Dynamics of Density-Stratified Reservoirs.'' In spite of the knowledge of the existence of density currents for many years, and the current research being done in this area, further research is needed. This need can best be summarized by quoting from a report entitled ''Internal Density Currents Created by Withdrawal from a Stratified Reservoir,''[110] a cooperative study of the Tennessee Valley Authority and the U.S. Corps of Engineers. They state:

> ''The need for knowledge of the character of internal density currents in reservoirs has become increasingly important in recent years because of the intensified multi-purpose use of such projects. Some of the uses of many reservoirs can only be optimized by knowledge of, and in some instances control of, internal currents. A few examples of the problems involved are: need to maintain cool, potable, domestic water; need for the coolest water possible for cooling thermal-electric power generation plants; need for maintaining water quality suitable for recreation, and for fish and wildlife; and need for manipulating water supplies to retard undesirable aquatic growth.
>
> ''Internal density currents have been studied by theoretical approaches, laboratory experiments, and direct measurement of velocities and stratification in prototype reservoirs. However, no clear-cut procedures have been made available for estimating in advance the internal density currents that might be expected in a proposed reservoir or for determining how such currents might be controlled. Reliable field data on velocities, temperature and density variations in deep reservoirs, are very meager. In addition, most of the theoretical studies and laboratory experiments have been only vaguely applicable to reservoirs in which densities often vary quite gradually with depth.''

Perhaps the sensitive Savonius Rotor Current Meter (measures to 0.017 ft/sec) or the Deep Water Isotopic Current Analyzer (range 0.0034 to 17 ft/sec) developed by the William H. Johnston Laboratory, Baltimore, Maryland, will prove useful in density current studies.

ARTIFICIAL DESTRATIFICATION OF IMPOUNDMENTS

Existing Information

Natural mixing and wind-induced currents have been studied in impoundments. Likens and Hasler[111] in ''Movement of Radiosodium in a Chemically Stratified Lake'' used sodium-24 to demonstrate horizontal movement of about 16 to 18 meters per day in all directions, but found no vertical movement. Lathbury, Bryson, and Lettau[112] in ''Some Observations of Currents in the Hypolimnion of

Lake Mendota'' showed that hypolimnion currents are very slow and are caused by a redistribution of density by the wind. Although currents do exist, they are too feeble to cause aerated water to penetrate into the hypolimnion and prevent the deterioration of water quality there. If destratification is to be accomplished, it must be done artificially.

The concept of mixing large bodies of water is not new; the first attempt was reported in 1952 by Hooper, Ball, and Tanner[113] in their article entitled ''An Experiment in the Artificial Circulation of a Small Michigan Lake.'' In the years between then and now (1964), at least 18 attempts (one on a model basis) have been made. Most of these have caused an improvement in the water quality of the impoundment or lake. Several different methods have been used, under different conditions and for different periods of time, but two features seem to stand out: (1) in a short time after the start of the operation, the degree of stratification was markedly reduced or eliminated, in most of the cases, and (2) the amount of equipment required was modest, considering the volume of water mixed.

Most authorities agree that the prevention of the deterioration of the hypolimnion through impoundment destratification would be beneficial. Mixing would prevent: (1) low dissolved oxygen concentrations; (2) increased iron and manganese concentrations, (3) production of hydrogen sulfide, and (4) increase in color, in the hypolimnion. Mixing would, however, prevent the accumulation of cool waters in the impoundment bottom and might increase overall productivity of the lake by recycling back into the euphotic zone nutrients released during organism decomposition.

Relatively small bodies of water have been mixed in the past, the largest reported in the literature being near 13 billion gallons (about 40,000 acre-ft). Derby[89] in 1953 reported on mixing the 13.2-billion-gallon lower Hollywood Reservoir with a diffused aeration system. He noted that this aeration lowered the thermocline, but did not completely prevent stratification. In 1954 Mercier and Gay[114] reported on the mixing of Lac de Bret, with a capacity of 13.1 billion gallons, by pumping from the bottom water and discharging the pumped water at a depth of 45 feet, 600 feet away from the pump. They noted that this mixing kept the lake 97 percent saturated with oxygen during one summer. Bryan[115] in his article ''Improvement in the Quality of Reservoir Discharges Through Reservoir

Mixing and Aeration'' reported on the mixing of Inniscarra Reservoir, which has a volume of 12.5 billion gallons, through the use of Aero-Hydraulic Corp. patented equipment, a large water pump with an air piston. The dissolved oxygen concentration only fell from 9 milligrams per liter in May, when the mixing started, to 3.5 milligrams per liter in July during 1961, while it fell to zero over the same period the previous year. This happened even though the rated pumping capacity of the equipment was only 20 million gallons per day. As was hoped, extra water was entrained during operation of the equipment.

While some attempts at mixing were more successful than others, only one experiment of the 18 described in the literature was not successful. Patriarche[116] in an article entitled ''Air-Induced Winter Circulation of Two Shallow Michigan Lakes'' tried to mix two shallow lakes with compressed air pumped in through a perforated rubber hose. He found this treatment ineffective in preventing depletion of dissolved oxygen, and the temperatures of both the water and the bottom muds were lowered. Since this technique has been quite successful in other lakes, the cause of these problems is not known. In this case fish mortalities occurred, and Patriarche felt that the air-induced circulation helped to cause these deaths instead of preventing them.

Research Needs

The research needs in this area fall into three categories: (1) to determine whether artificial circulation of larger storage impoundments is technologically feasible; (2) if this is possible, to determine the most economical means and cost of circulation; and (3) to determine the total effect of circulation on the water quality in an impoundment.

Wherever the field investigation is done, it must encompass a comparison of the two or three major methods of moving large quantities of water through a low head. These methods must be compared on the basis of economics, as well as performance. While the tests are being made, sufficient measurements must be taken to evaluate the total effects on the water quality, both beneficial and adverse. Finally, such factors as the best time to start mixing and the duration of water movement after deactivation of the power source must be evaluated. To obtain some preliminary information on artificial destratification, the Engineering Section at the Robert A. Taft Sanitary Engineering Center has assembled a 2,900-gpm, raft-mounted low-head pump for drawing cold bottom water to the surface where it is discharged. Initially this equipment will be tried on small, 30-million-gallon lakes. If this operation is initially successful, destratification will be attempted in a larger lake of 0.5 billion gallons. Mixing of large volumes of water may be impractical or detrimental, or it may be a very worthwhile solution to many of the present problems in impoundments. Only research will settle the issue.

EVAPORATION CONTROL

Existing Information

Most of this report deals with water quality characteristics and criteria, but streamflow regulation may also be undertaken to help meet a downstream water quantity criteria. As Neel has pointed out in an unpublished report, whenever an impoundment is constructed to regulate streamflow, the flow variation in the stream may be smoothed out, but the total annual flow will be reduced, except in humid areas, because of increased evaporation from the impoundment. Since this is the case, and downstream water quantities are important, the possibility of reducing evaporation becomes an important consideration. One estimate by Meyers[117] in ''Evaporation from the 17 Western States'' gives the annual loss from impoundments in these western states at 15,600,000 acre-feet per year, but the Senate Select Committee sets the loss at 21,000,000 acre-feet per year. Recently an American Water Works Association group, Task Group 2480R - Evaporation Control, with Curtis Michel as chairman, reported on ''Survey of Methods for Evaporation Control.''[118] This is a very useful review of the entire subject of evaporation control and contains 54 references.

This group suggests seven important considerations in the field of evaporation control:

1. Siting - ''Of all the methods that have been suggested to control evaporation from reservoirs, site location is perhaps most important. If a reservoir can be situated to reduce the effects of shallow areas, high winds, and perhaps even excessive solar radiation, evaporation losses can be reduced.''

2. Shallows - ''Avoiding extensive areas having shallow water depth should help to curtail excess evaporation losses. The costs of the fill or diking and maintenance would have to be considered for each situation. ... Areas shallower than the depth of annual evaporation contribute little to useful storage and may serve to increase salinity.''

3. Phreatophytes - ''Non-beneficial, water-consuming plants, phreatophytes, are generally regarded as a nuisance. These will be found mostly in the shallows and along the shoreline. But before they are removed, the possibility of increased erosion should be considered as well as whether the costs of removal and possible hazards to aquatic life and ecological balance of the reservoir or lake are justified by the savings.''

(Note the Senate Select Committee[3] estimates a yearly loss of 22,000,000 acre-feet by phreatophytes in 17 western states.)

4. Wind breaks - ''Similarly, wind breaks must be considered in the light of their net contributions to water savings. Obviously trees and plants consume water. ... One report states that tree planting along lake shores can reduce winds to leeward up to a distance of 20 times their height. Whether the reduction in evaporation that a somewhat reduced wind force will effect is greater than the water requirements of the planting is not clear.''

5. Covers - ''Floating covers, consisting of reflective material such as white polyethylene spheres, have been used on oil storage tanks and tested on water storage. Cost and interference with other reservoir uses limit this method, while capillary action is liable to render it ineffective under slight wave action. Solid plastic covers are similarly impractical for large storages and interfere with gaseous exchange. Roofing reservoirs has been suggested and the costs have been estimated. The costs appear excessive and the installation would conflict with the various

uses to which reservoirs are put, in particular with the role as fish and wildlife preserves and with the increasing recreational use.''

6. Seepage - ''Seepage, while not properly within the discussion, is a reservoir water loss that should be mentioned. On large water storages, reduction of seepage by treatment with soil stabilizers probably costs more than the value of the water that may be saved though this appears to be a feasible method for ponds and canals. ... As an impoundment ages, seepage losses are reduced.''

7. Monolayers - ''It may be inferred from the discussion that there are two ways to reduce evaporation: (1) Decrease the surface area of the water exposed to the air, or (2) lessen the effects of the physical forces that cause evaporation. The method of controlling evaporation by the application of a monomolecular film may be regarded as doing both, by damping the microwaves and imposing an additional physical resistance.''

In this article, and in research through the years, the greatest attention has been given to the possibility of evaporation control by covering large impoundments with monomolecular films, using either normal hexadecanol or normal octadecanol, to suppress evaporation. Under ideal conditions, these films are remarkedly effective in retarding evaporation. But in a field situation, because of such adverse conditions as wind, waves, bacterial degradation, molecular holes arising from incomplete evaporation of solvents, entrapped foreign materials such as dust, and inability under certain conditions to maintain film compression, the effectiveness of this treatment has been reduced. Most of the field trials have shown very modest results, although the Task Group admits that it is very difficult to truly measure the effect of these films in a field situation. Another very useful reference on this subject is a collection of 19 papers presented at a symposium on September 15, 1960, titled ''Retardation of Evaporation by Monolayers.'' The work is edited by Professor Victor K. LaMer.[119]

Research Needs

Investigations in this area have been both promising and disappointing. Control of this enormous water loss is too important to neglect, however. Therefore, research efforts refining the hexadecanol method and seeking an entirely new approach should continue. It is of interest to note that only one Research Grants Branch project on this subject, ''Evaporation Suppression and Its Effects on Biota,'' Grant Number WP-00160, supervised by Franzini, could be found in their 1963 List of Awards.

THERMAL POLLUTION

Existing Information

The addition of warm water to an impoundment, either because of the warming of the influent waters because of solar radiation or because of the discharge of warmed cooling water from industrial processes, influences the water quality in impoundments. These warm waters tend to stay near the surface of the impoundment. This warm water influences aquatic life in the surface waters of the impoundment, influences the rate of deoxygenation from biological degradation of organic matter by speeding the process, and influences dissolved oxygen concentrations because of decreased solubility of oxygen in higher temperature water. The latter reduces the surface deficit, and thereby reduces the reaeration rate.

Thermal pollution of streams has received some attention in the literature. Hoak[120] in ''The Thermal Pollution Problem,'' Arnold[121] in ''Thermal Pollution of Surface Supplies,'' and Churchill[122] in ''Control of Temperature Through Streamflow Regulation'' are some of the authors who have given attention to this problem. The first two articles discuss problems of the danger to aquatic life and of lowered dissolved oxygen concentrations caused by warmer waters. In addition they suggest criteria for avoiding problems. Hoak[120] refers to the work of the Pennsylvania Sanitary Water Board and quotes the regulations for the Commonwealth on stream water temperature. Duttweiler[123] in ''A Mathematical Model of Stream Temperature'' has developed a model, which works reasonably well in a small stream and can be used to predict downstream temperatures below a streamflow regulation release.

Churchill's article is more concerned with the influence of discharging colder hypolimnion water on downstream water temperatures. He shows that the water temperature can drop abruptly from a river water temperature of 75°F down to the hypolimnion temperature of 45°F when large discharges of hypolimnetic water are made from an impoundment. This large a temperature change places considerable stress on downstream aquatic life.

One article found discusses in detail thermal pollution in impoundments; Eckel[124] in "Temperature Condition in Impounded Waters" discusses conditions of uptake, distribution, and discharge of heat in flowing and impounded waters and the differences between natural lakes and impounding reservoirs. These differences are illustrated in this German article by records of measurements in a large number of lakes and impounding reservoirs.

The Proceedings of the Twelfth Pacific Northwest Symposium on Water Pollution Research[125] entitled *Water Temperature, Influences, Effects, and Control* is an important review of this thermal pollution problem, as is indicated by some of the papers included, which are listed below:

1. "Temperature as a Water Quality Parameter" by C.M. Everts.
2. "Effects of Water Uses and Impoundments on Water Temperature" by R.O. Sylvester.
3. "Temperature Phenomena and Control in Reservoirs" by J.M. Raphael.
4. "Some Observations of Columbia River and Reservoir Behavior from Hanford Experience" by R.T. Jaske.
5. "Instrumentation for Water-Temperature Studies" by A.M. Moore.
6. "Summary of Current Theories and Studies Relating to Temperature Prediction" by R. Zellar.

Jaske agrees with Churchill[122] that water control structures can have a profound influence on temperature and, if designed and used properly, can promote desirable downstream conditions while avoiding undesirable ones. This reemphasizes the need to understand the influences caused by the design and operation of outlet works.

Research Needs

While the problem of thermal pollution in streams has been investigated and discussed in the literature, little information concerning these same effects in impoundments has been published. Addition of warmed water to an impoundment may well affect its behavior, and this effect must be understood to gain complete understanding of imments. These studies should be on a field scale since studying this problem on a laboratory scale would be difficult. A large, well-instrumented model impoundment containing typical aquatic life might be used, however, to gain some insight into the influence of the introduction of heated waters. The blending of warm and cool waters is a problem in mixing, and the references noted in the section on Turbulent Diffusion and Natural Mixing may be applicable here.

Summary of Russian Experiences

Reference is made to a Russian article, ''Prognosis of Reservoir Quality.'' by Chibov.[126] This article is of particular interest since it summarizes the state of knowledge of reservoir behavior as understood by Russian investigators. A review of various abstracting journals has revealed that there is a great deal of interest in Russia, Czechoslovakia, and Yugoslavia on the general topic of water quality behavior in impoundments.

In Chibov's article an equation is given for the calculation of the changes in mineral content of an impoundment between seasons T and seasons T + 1. The author is enthusiastic about the formula, which takes the form of a mineral balance in a given impoundment, but is critical of the fact that the studies on which the formula was based were all made in impoundments near one geographical location. He also points out that merely being able to forecast mineral content does not truly give a complete picture of the effects of reservoirs on water quality. He suggests that physical, chemical, bacteriological, and other biochemical aspects of water quality must all be understood in an impoundment environment. The author is also critical of members of the VODGEO Research Institute, who developed the previously mentioned prediction formula, in the following statement. ''In the first, they make no reference to river basin classification; they disregard the effects of local conditions, such as river slope, shape of bed in either sectional direction, size of the flooded area above and to either side of the dam, the rate of water flow, direction of prevailing winds, etc. Their approach to the problem is of an abstract character and has no direct bearing on concrete cases.''

From this quotation and the tone of the remainder of the article, it would appear that although intensive studies have already been made in the USSR, additional work is needed in order to develop a complete scheme that will allow the understanding of the influences of impoundments on water quality. This thought is echoed in a conclusion in the U.S. Public Health Service's report on the "Pearl River Shallow Water Reservoir Water Quality - Recreation Project"[35] which states, "While some water quality changes such as turbidity and bacterial reduction and temperature stabilization may be expected to result from impoundment, it is difficult at this time to make further predictions, i.e., algal growth potential, manganese content increase, etc."

INFLUENCE OF IMPOUNDMENT RELEASES ON DOWNSTREAM WATER QUALITY

Although the major emphasis in this report relates to the effects of impoundments on water quality, the influence of impoundment releases on downstream water quality must not be overlooked since impoundments are merely a portion of any streamflow regulation project. The subject of stream mechanics and the behavior of water quality in flowing waters has received considerable attention through the years. A complete review of this subject would be an extensive report in itself. The following discussion is limited to a review of the existing information on water quality in streams as influenced by impoundment releases.

INFLUENCE ON QUALITY

Existing Information

Very little has been written on the influence of impoundment releases on downstream water quality, and those articles that have been prepared are concerned with the degradation of downstream water quality because of releases of poor quality water from impoundments. Ingols,[66,67] discusses the downstream stress caused by discharge of water with a low dissolved oxygen concentration and a high manganese concentration during peaking power operations. He realizes that water with a low dissolved oxygen concentration reaerates rapidly because of the high oxygen deficit, but he reports the absence of fish life in stretches below peaking power dams. Although TVA investigators[127] have found fish under these circumstances, problems with downstream water treatment because of manganese concentrations, and difficulty in using the stream for the assimilation of organic wastes under these conditions have to be resolved. Ingols also mentions that sustained slow releases of this low quality water would be detrimental, but releasing it as a power wave aggravates the situation.

Hall,[128] in his article ''Reregulation of Impounded Waters, Chattahoochee River, Georgia'' recognizes the problem of power wave discharge and discusses the possibility of a secondary impoundment downstream (reregulation) from the major peaking power impoundment to smooth out the peaking power releases. Fish,[129] in his article ''Effect of Impoundment on Downstream Water Quality, Roanoke River, N. C.,'' discusses many of the problems mentioned above as they are related to the Roanoke River situation. He too notes the degradation of downstream water quality attributable to hypolimnion releases from impoundments.

The Tennessee Valley Authority has made several studies on this problem through the years. Churchill, in a talk entitled ''The Effects of Reservoirs on the Water Quality,'' presented at the National Council for Stream Improvement's Annual Meeting, February 18-21, 1963, discussed one situation in which extremely high manganese concentrations developed in the bottom waters of an impoundment. He noted that these concentrations were so detrimental to downstream water quality that bottom water could not be drawn from this reservoir. In the paper mentioned previously, Churchill[122] discusses the influence of withdrawals from deep impoundments on downstream water temperature.

Pfitzer[127] discusses the influence of impoundment discharge on fish populations in an article entitled ''Investigation of Waters Below Storage Reservoirs in Tennessee.'' He states:

> ''These various chemical and physical changes have caused radical alterations in the plant and animal life. Two very striking examples show the effect on the fish populations. An unpublished preimpoundment survey in 1947 by Charles J. Chance and Reeve M. Bailey in the South Fork of the Holston and Watauga Rivers recorded 43 species for South Fork and 32 species for Watauga. After construction of the dam and subsequent discharge of cold water, only 17 species of fish have been recorded for South Holston tailwater and 13 for Watauga and Wilbur tailwaters on the Watauga River.''

Research Needs

The literature cited above is concerned with the detrimental effect on water quality because of the discharge of poor quality water from impoundments. No discussions were found concerning the beneficial effect that might accrue from the discharge of good quality water. Yet, this is precisely the goal of a streamflow regulation project, the discharge of high-quality dilution water that will have a beneficial effect downstream. One might presume that if the dilution water were of good quality, the effect on a given water quality characteristic downstream could be calculated on a weighted proportion basis, taking into account the relative concentrations of the substance under consideration and the relative volumes of the downstream water and the dilution water. For substances that do not undergo reactions, such as chlorides, this approach is probably valid, but it will not necessarily work for all substances.

Several effects need to be investigated. Examples of these are as follows:

1. The influence of dilution on rate of biological deoxygenation, must be investigated. The dilution of a given polluted stream may cause an increase or a decrease in the rate of biological oxidation of the organic matter in that stream.
2. Because of increased turbulence and increased depth caused by increased flow, the rate of reaeration may be changed in a given situation when dilution water is added.
3. Because of increased flow, the time of travel between a source of pollution and a possible water use will be shortened. In certain situations poorer quality water may arrive at a downstream point because this increased time of travel would decrease the opportunity for natural purification to occur.
4. If algal toxins are present in the river, and these toxins are preventing an undesirable bloom of algae, the dilution of this water with toxin-free water from an impoundment may well result in undesirable effects.
5. The discharge of good chemical quality, but low temperature, water may interfere with downstream recreational facilities or the type of fish present.

6. The absence of fish life below some dams discharging hypolimnion water, even though it has been reaerated, has led to the supposition that in the reducing environment of the hylolimnion, substances toxic to aquatic life are produced. If this possibility is true, it must be investigated since even on reaeration, this water may be undesirable as dilution water.

These are six areas in which additional information is needed badly if the complete effect of a streamflow regulation scheme is to be assessed. Many of these studies can be carried out in the laboratory by blending various concentrations of river water with dilution water of a given quality and the resulting behavior of the mixed waters determined. Extreme care must be taken when making stream studies under laboratory conditions, however, because a field situation is so much more complex and probably never can be actually duplicated in the laboratory. Other problems, such as the influence of decreased time of travel, may better be studied in a controlled field situation. In summary, the influence of impoundment releases on downstream water quality is a challenging but potentially fruitful and necessary area for continued investigations.

TURBULENT DIFFUSION AND NATURAL MIXING

Existing Information

An adequate knowledge of mixing phenomena in rivers is essential to assure the most efficient use of dilution water. If dilution water becomes channeled and does not mix with the entire contents of a river, the entire benefit of this water will not be realized. This problem would be accentuated in wide, relatively slow-moving rivers where waste inputs as well as dilution water might cling to one shore or the other and might be very slowly blended. Wisniewski[130] in ''Cross-Section Surveys as a Method of Evaluating the Self-Purification Capacity of a Stream'' reports on a situation in which an effluent remained on the same side of a river as the original discharge for 12.5 miles downstream.

The mathematical models needed to describe turbulent diffusion are complex and only recently have been attempted by various investigators, previous studies being made with the help of tracers as described by Thomas and

Archibald[131] in ''Longitudinal Mixing Measured by Radioactive Tracers.'' Currently, important research on this problem is being carried out at Vanderbilt University under the direction of Krenkel. As the title of his U.S. Public Health Service project, WP-00328, Turbulent Diffusion and River Waste Assimilative Capacity, indicates, he is interested in turbulent diffusion as it influences reaeration (Krenkel and Orlob[60]), but his research will also give valuable insight into mixing phenomena.

To review this subject, portions of Krenkel's project. progress report[132] are quoted below.

> ''(1) Eddy Diffusion
>
> ''Oceanographers and meteorologists have long used the analogy between diffusion of matter on a microscopic scale and the spread of a tracer material on a macroscopic scale to describe turbulent mixing in the ocean and in the atmosphere. This process can be described by Fick's second law, which is
>
> $$\frac{\partial C}{\partial t} = D_E \frac{\partial^2 C}{\partial x^2} \qquad (2)$$
>
> in which C is the concentration, t is the time, x is the distance and D_E is the eddy diffusion coefficient. Although the analogy has been subjected to question, one can find much experimental evidence of the molecular diffusion process on a macro-scale as applied to the spread of tracers in a turbulent field both in the ocean and in the atmosphere. An excellent summary of the process, including experimental verification of the analogy, can be found in a report by Harleman et al. (1).
>
> ''Although equation 2 has been used successfully in numerous diffusion investigations, it should be recognized that in the case in question, (that is, in open channel flow), use of this mathematical model may result in erroneous conclusions. The concentration gradient predicted by this model may be in error because of the neglected mass transport caused by the convective velocity. Also, the eddy diffusion coefficient as previously defined describes mixing at a point in the flow, and as such may not be the parameter desired to denote mass transfer in

a stream section. To predict the mixing characteristics of a river section using equation 2, it would be necessary to determine the eddy diffusion coefficient at numerous points and then use an appropriate averaging technique to obtain the overall description of mixing in the section under consideration. As shown subsequently, this averaging process is performed automatically when the longitudinal mixing coefficient is used.

''(2) Longitudinal Mixing Coefficient

''Consider an experiment in which an instantaneous dose of tracer has been injected into an open channel where the water is flowing at a mean velocity, $\bar{u}$. If samples are taken with respect to time at a station downstream from the point of injection, the flow pattern will typically follow equation 1:

$$\frac{\partial C}{\partial t} = D_L \frac{\partial^2 C}{\partial x^2} - \bar{u} \frac{\partial C}{\partial x} \tag{1}$$

which is the general differential equation describing the mixing process of the injected tracer. A solution of this equation is

$$C = \frac{M}{A(4\pi D_L t)^{1/2}} \exp\left[-\frac{(x - \bar{u}t)^2}{4D_L t}\right] \tag{3}$$

where M is the weight of injected tracer material, A is the cross sectional area and x is the distance downstream from the point of injection.

''Equation 3 has been found to describe very closely the temporal distribution of an instantaneously injected tracer, and is similar to the equation developed by Taylor (2) (3), partly by intuition. Examination of this equation shows it to be representative of the normal error curve if the tracer distribution with distance is observed at a particular time. Normally, in experiments of the type described, the tracer is instantaneously injected and the concentration distribution as a function of time is observed at a fixed station downstream. Obviously, because the exponential denominator in equation

> 3 is no longer constant in this case, one would expect to obtain a skewed time-concentration distribution. This observation has been made by Taylor who states that the asymmetry should be negligible when the observation point is greater than 100 pipe diameters downstream from the point of tracer injection. A physical explanation of the skewness may lie in the fact that diffusion is taking place while the tracer cloud is passing the observation point.
>
> ''It should be noted that when a significant laminar sublayer exists, the time-concentration curve may not follow equation 3 because of the slow release of tracer particles from the sublayer itself. This could explain the extremely long tails that are obtained in tracer tests conducted in sedimentation basins, for in this case it is highly probable that a significant laminar sublayer is present.
>
> ''Elder (4) in an attempt to explain the deviation of the observed time-concentration curve that is sometimes obtained from the theoretically predicted distribution, proposed the following mechanism. The observed time-concentration curve is considered to be composed of two components – the primary one corresponding to tracer that is carried in the turbulent flow regime, and the secondary one resulting from the slow release of tracer existing in the viscous sublayer. Elder considered that particles contained in the laminar layer could only be released to the turbulent portion of the stream via the process of molecular diffusion.''

Note, the references are for those contained in the original report, which contains a useful bibliography containing 20 references on turbulent diffusion.

Krenkel concluded his literature review on mixing by saying:

> ''Perusal of the literature has demonstrated that although several investigations have yielded information on a particular river under an existing flow condition the data necessary to predict and understand the diffusion phenomenon are not available. It is obvious that in order to more fully delineate the parameters

> involved in the overall mixing process in inland waterways, controlled data from field scale measurements are required."

Research Needs

Current research needs in this area are being attacked by Krenkel, who notes the following objectives of his project.

> "The specific objectives of this investigation were as follows:
>
> "(1) To determine the applicability of the mathematical model describing turbulent diffusion with convection in open channel flow:
>
> $$\frac{\partial C}{\partial t} = D_L \frac{\partial^2 C}{\partial x^2} - \bar{u}\frac{\partial C}{\partial x} \qquad (1)$$
>
> in describing the concentration gradients for a point or line source of waste contaminant in two-dimensional flow.
>
> "(2) To determine the relation between equation (1) and the conventionally defined eddy diffusivity, D_E:
>
> $$\frac{\partial C}{\partial t} = D_E \frac{\partial^2 C}{\partial x^2} \qquad (2)$$
>
> in a homogeneous turbulent field.
>
> "(3) To examine the methods of determination of these mixing coefficients in the model and to determine the practicability of these methods for field scale studies.
>
> "(4) To evaluate diffusion and longitudinal mixing coefficients for artificial and natural channels in terms of conventionally used hydraulic parameters.
>
> "(5) To correlate turbulent diffusion characteristics of natural streams to the reaeration capacity and thereby the waste assimilative capacity.

> ''(6) Utilizing pulse testing techniques, to develop the frequency response characteristics of various streams under differing hydraulic conditions.
>
> ''(7) To test the model and techniques developed as a result of these model studies on the prototype, i.e., field scale studies on a natural river.
>
> ''Although these objectives appear to be quite optimistic, a logical solution to the overall mixing process in inland waterways requires that each be subjected to analysis. Also the experimental procedures may be designed such that information pertinent to each phase of the investigation will be obtained.''

The progress of his work is briefly summarized below.

> ''Because the initial phases of these investigations were largely involved with equipment design and refinement of methodology, the major portion of the experimental data will be obtained in the ensuing months. It is expected that sufficient data will be produced by October of 1964 such that the desired analyses can be performed, both in the laboratory flume and in the selected TVA rivers.''

OPERATIONS RESEARCH FOR WATER QUALITY MANAGEMENT

MATHEMATICAL ANALYSIS

Existing Information

The mathematical analysis and optimization of complex streamflow regulation systems is a relatively new field, but a vital one. Work in this area is being pioneered by Thomas and his co-workers at Harvard University and Kneese with Resourses of the Future, Inc., among others. In their final report on ''Operations Research in Water Quality Management,'' U.S. Public Health Service Project Number WP-00111, Thomas and Burden[133] have attempted to show how the ability of various engineering alternatives to accomplish a given objective can be represented in mathematical terms. These are called technological functions. Once this has been done for a given situation, they employ techniques for relating the effect of several engineering alternatives and then optimizing the entire system. The mathematics needed to accomplish this are, of course, complicated and sophisticated, and Thomas has had to develop new mathematical approaches to overcome gaps in the state of the operations research art, but progress has been made and is continuing in this area.

Further progress is essential since this is the only known method of optimizing the design of a complicated streamflow regulation project. Such a project might involve impoundments, rivers, water treatment and wastewater treatment plants, industries, hydro and thermal power plants, uncontrolled pollutional discharges, recreational areas, flood control projects, and transportation and irrigation systems in any number and combination. Each of these contains several design as well as operating variables, each of which in turn influences the downstream water quality at some given point. The complexity of even a small river basin such as the Lehigh Basin, which has been studied by Thomas[133] and his group, is enormous. Only through an operations research approach, however, can the best design and operation of such a river basin facility be accomplished.

All of the sophisticated scientific knowledge that may be amassed eventually to allow the understanding and prediction of impoundment and stream behavior will be of maximum benefit when this knowledge can be integrated into an optimum design for an overall river basin. One example of the current use of available systems analysis techniques, despite their shortcomings, is in a paper by Thomann,[134] "Use of Systems Analysis in Estuarine Water Pollution Control." Good agreement was obtained between predicted and actual dissolved oxygen concentrations at Torresdale, Pennsylvania.

Progress in this area is important from another point of view. A mathematical analysis of a design often gives insight into the most sensitive variable in the design. For example, in the design of a streamflow regulation project, changing each of the variables in the equations independently reveals which variable change causes the greatest change in the overall solution to the problem. Once the most sensitive variable has been determined, the most benefit is gained by emphasizing research effort related to this factor. Through this process the major research effort is placed on the most critical variable.

The growing interest in the field of the use of mathematical analysis for the solution of water quality problems was indicated by the attendance of 24 persons at a 1-day symposium, April 6, 1963, organized by Dr. Richard L. Woodward, formerly of the U.S. Public Health Service. This informal gathering followed the Symposium on Streamflow Regulation for Quality Control, April 3-5, 1963, sponsored by the U.S. Public Health Service in Cincinnati, Ohio. The purpose of this meeting was to bring the researchers in this area together to meet one another and to exchange views. At the lively all-day discussion the concensus was that this is a vital area in which vast amounts of additional work are needed.

Research Needs

The research needs in this area involve both mathematics and engineering. New mathematical techniques are needed to further refine existing capabilities. More refined engineering data are needed to be used in the mathematical formulations. An example of the latter need may be found in chapter 3 of ''Operations Research in Water Quality Management,''[133] which is titled ''Queueing Model for Pollution Transport in Streams.'' This chapter is concerned with the development of mathematical techniques for the formulation of the complex process of pollution transport in streams. Some very sophisticated mathematics for accomplishing the above mentioned objective have been developed, but the present weak point in the method is the understanding of such items as: the mechanism for the degradation of organic wastes, the role of bottom deposits, and the role of photosynthetic oxygen production in the overall oxygen balance. Only further engineering research can strengthen these points.

As mentioned previously, the mathematician can often indicate the variable that requires the most research, that is, the one upon which the final result is the most dependent. The mathematician, however, must recognize that his current data may be insufficient for establishing the most sensitive variable in a water quality management problem. Information and research, therefore, must flow in both directions. Engineering research must provide improved information on scientific phenomena to the mathematician so that he can refine his mathematical analyses, and feed back to the researcher a better estimate of where additional work needs to be done. This continual exchange of information requires additional education of both groups, and is essential if systems analysis is to become a truly beneficial tool.

ECONOMIC ANALYSIS

Existing Information

Cost/Benefit Ratio. Most of the writings on this topic are concerned with the problem of evaluating the economic benefits provided through the regulation and management of some particular water resource, such as providing a certain volume of water in a particular location or preventing flood damage. The losses that occur through failure to meet a water quality or quantity objective, or as an associated effect of providing other benefits, are also involved. In the situation where additional storage capacity for water supply is provided in a multi-purpose impoundment, the economic benefits have, in the absence of any better method, traditionally been calculated as equal to the cost of providing an equivalent supply from the next most feasible source in the absence of the project. This may be a single-purpose impoundment, a well supply, or a supply purchased from some other system already in existence. In the case of storage provided for the purpose of regulating a stream for water quality control, benefits were formerly computed on the basis of the cost of removing a pound of BOD from the stream. For several reasons this concept has been abandoned, and the method now applied is the same as that used in evaluating municipal water supply benefits. The use of this method in this case, however, is even more questionable than in the case of water supply, but in the absence of anything better must be relied upon. Current research by economists and mathematicians has as its objective the development of a procedure that will yield a more realistic monetary evaluation of benefits attributable either to the provision of municipal and industrial water or to the maintenance of water quality.

Chapter 7, entitled ''Imputed Benefits of Municipal Water Supply,'' of ''Operations Research in Water Quality Management''[133] describes a method of assigning a value to municipal water supply and presents the mathematics necessary to handle a small example. This method is based on determining the increase in monetary benefits that would occur if the water supply requirement was reduced by one unit and this unit was diverted to an alternative use that has a calculable monetary benefit. For a given situation, an imputed benefit for the municipal water is thereby determined.

The book *The Value of Water in Alternative Uses*[135] takes a somewhat different approach. In the example cited, a given quantity of water is to be distributed between the San Juan and Rio Grande basins in New Mexico. The object of this analysis is to determine the most beneficial distribution and ultimate uses of this water. The possible uses are irrigation, municipal-industrial water supply, and recreation. Different volumes of water are assigned to each of these uses, and eight models are analyzed economically to determine which model would increase the gross product of the state the most. This analysis is made by calculating the increase in revenue that would come into the state for each particular combination. Through this technique the most beneficial division of this San Juan river water was determined.

In "The Animal Farm" Thomas[136] addresses himself to the problem of the determination of a mathematical model for the discussion of quality criteria for control of the environment. In this article he suggests that every quality criterion established contains an implied cost/benefit ratio and that the cost/benefit ratio should be taken into account when criteria are selected for the model.

Still another approach is contained in the report "Economic Evaluation of Water, Part III, An Interindustry Analysis of the California Water Economy" by Lofting and McGauhey.[137] This analysis is based, in general, on the Leontief analytical system, which is often known as the input-output method. This method involves constructing an input-output table in which output of each industry is distributed among other industries as their input for further processing. In this study, the average water use pattern for California industries is superimposed on an approximate interindustry transaction matrix for the same period. By this method water uses and requirements are correlated to the general flow of money and product through the large economic system. For example, a principal user of water is agriculture, and a number of industries make fairly heavy demands on agriculture for their product input; continual inputs of water to agriculture are therefore necessary to maintain the economic health of the dependent industries.

These four examples give some insight into the mathematical approaches being used to attempt to evaluate costs and benefits for water resource systems. The classic reference in this area is *Design of Water Resource Systems* by Maass, Hufschmidt, Dorfman, Thomas, Marglin, and Fair.[138]

This book, popularly known as ''The Harvard Blue Book,'' is the culmination of 5 years work by the personnel of the Harvard Water Program.

Recreation. Methods for calculating the economic benefits that realistically result from provision of facilities for outdoor recreation have been given some attention by economists. The rapidly increasing demand for water-oriented recreation, stemming from such factors as increases in population, leisure, incomes, mobility, and urbanization, prompted creation of the Outdoor Recreational Resources Commission by Public Law 85-470 in 1958 to determine the nation's needs in this area up to the year 2000. The economic gains from the provision of these facilities must be evaluated if the correct cost/benefit ratio for a water resources system containing recreational facilities is to be determined. Several methods are available for attempting this calculation, although each has its deficiencies.

In gathering information for *The Value of Water in Alternative Uses*, Wollman[135] circulated questionnaires to sportsmen and recreationists in an attempt to determine the amount of money that the users of a recreational facility spend. These costs include items such as the cost of gasoline to make the trip, the cost of equipment used at the site, and the cost of food consumed at the site. Similar expenditures per acre of recreational facility or mile of stream are then presumed to be applicable to new recreational facilities in the area. The size of a new facility would be a measure of the increased flow of money generated by the new facility.

Clawson[139] in her paper ''Methods of Measuring the Demand for and Value of Outdoor Recreation'' uses a somewhat different approach. Her method makes use of a demand curve for a recreational area. The method relates the cost of a visit to number of visits (the lower the cost, the greater the visits) and uses this relationship to determine the economic worth of the recreational area. Revenues obtained from monies spent inside a park decline when entrance fees are charged because of decreasing visits. The relationship between entrance fees and number of visits permits determination of the optimum entrance fee for maximum income. The value of the resource is then taken as the capitalized net profit resulting from imposing the most profitable entrance fee. This is a useful approach, but has limitations because of simplifying assumptions. These are discussed by Knetsch[140] in his paper ''Outdoor Recreation Demands and Benefits.'' This article takes into account other factors such as increased land values in close proximity to a recreational area and the effect of time of travel to a recreational area.

The perfect method of evaluating these real and often high monetary benefits from recreational areas has not yet been determined, but improvements are being made continually.

Loss Functions. Some economists are giving attention to determining loss functions. A loss function is an expression of the method of calculating an economic or monetary loss that occurs when some water criteria is not met, whether it be a quality or quantity criterion. In industrial terms, this might be an increased production cost because of the necessity of treating process water more intensively at certain times of the year because of a change in some particular quality characteristic. In municipal terms, an economic loss might occur if an insufficient quantity of water at a particular time of the year causes water restrictions in a given municipality, restricts new home construction, or influences industrial expansion. Determination of these loss functions is important. If streamflow regulation projects could prevent an economic loss and the loss could be calculated, this amount of avoided monetary loss could rightfully be credited as a benefit to the project. This is a logical extension of the methodology applied to flood control or highway evaluation.

Specific references on this subject were not found, but Bower at the University of New Mexico (currently with Resources for the Future, Inc.) is determining loss functions for the particular industries in his area with respect to possible changes in water quality. Extensions of this work are necessary if the systems analysis approach is to be truly complete.

Research Needs

Research by economists, sociologists, and engineers in the area of economic analysis is needed to improve the economic evaluation of streamflow regulation projects. As was stated in the section on mathematical analysis, further scientific information is necessary to improve the data used in economic analyses. For example, in the important area of the development of loss functions, engineering research designed to develop new treatment techniques that will ensure the maintenance of a good quality finished water during periods of raw water quality deterioration is necessary. The construction and operating costs of plants that would use these new treatment techniques could then be integrated into the economist's loss functions, since the cost of this additional treatment would truly be an economic loss caused by failing to meet some raw-water quality objective. In some cases the failure to meet a water quality objective can be overcome by: (1) application or refinement of existing treatment methods, (2) use of an additional source of supply, or (3) use of an entirely new treatment scheme. Whichever alternative is needed, the engineer must be able to supply the economist with the proper alternative, and this cannot be done without further research in the area of water and wastewater treatment. If the failure cannot be overcome, a loss in income will result because of lower production or the cost of a shift to another activity.

In the economic and social area, not only must improved methods of manipulating the monetary values involved in water resources systems be developed, but also additional attention must be given to the social aspects of the problem. Considerable effort on studies of the motivation and behavior of the general public in the important area of willingness to spend tax money on water quality control systems must be made. The finest water resource system design, carefully optimized by mathematicians, economists, and engineers, may well be stymied because of an uninformed, unprepared, or disinterested voting public. Because of the competition for the consumer dollar, realistic methods for motivating people to spend tax money for these water quality control systems should be further investigated. Some of these problems were discussed in in ''The Social and Economic Aspects of Water Resource Quality Control''[141] but more effort is needed.

CURRENT STREAMFLOW REGULATION PROJECTS AND FUTURE ACTIVITIES

Current Projects

The amendment to Public Law 660, Public Law 87-88, was passed on July 20, 1961; review of what has been done in the intervening years to implement this law is pertinent to this report. The current state of affairs was effectively summarized by Eugene W. Weber,[142] Deputy Director of Civil Works for Policy, Office of the Chief of Engineers, Corps of Engineers, in his paper ''Relation of Regulation for Quality Control to Activities of the Corps of Engineers,'' presented at the Symposium on Streamflow Regulation for Quality Control, April 3-5, 1963, sponsored by the Robert A. Taft Sanitary Engineering Center. Portions of this paper are quoted below.

> ''The enactment of Public Law 87-88 on 20 July 1961, less than 2 years ago, has made it possible to give full consideration in the planning and development of the Corps of Engineers' water resources program to provisions [sic] for regulation of streamflow for water quality control purposes on a par with all other project purposes. I will outline briefly the import of this new authority on three categories of projects in the Corps' program: i.e., completed projects; those authorized, but not yet completed or started; and those being studied for possible future authorization.

COMPLETED PROJECTS

> ''The Corps, in 1962, had in operation 214 reservoir projects containing 169 million acre-feet of storage capacity, primarily for flood control and power purposes with provisions for recreational use at most and with storage for water supply at some. Needs for flow regulation for water quality control were considered in planning many of these projects, but due to the absence of a positive policy for Federal participation in providing for such regulation, specific measures for quality control were included in relatively few projects, such as Fall River, Toronto, and Kanopolis Reservoirs in Kansas; Berlin and Mosquito Creek Reservoirs in Ohio; Youghiogheny River Reservoir in Pennsylvania; and Tygart Reservoir in West Virginia.
>
> ''The importance of low flow augmentation has been amply demonstrated by these earlier projects. For example, the operation of Fall River Reservoir, Kansas, in the very severe drought in the 1950's provided vitally needed water supply and dilution of wastes involving six downstream municipalities and also prevented

complete shutdown of three refineries and other industries. As another example, the Berlin project, Ohio, was constructed in the early 1940's at the request of the War Production Board primarily for low flow regulation in the Mahoning River Basin to maintain maximum power and steel production in this highly industrialized valley. In view of the continued critical situation with respect to steel production, the method of operation specifically adopted as a war-time measure was continued until 1949 at which time a more balanced method of operation between flood control and low flow regulation was adopted. Some of our very early studies of low flow augmentation, which were carried out in cooperation with the Public Health Service as far back as the late 1930's, established the need for the inclusion of regulatory storage in projects only recently undertaken, such as the Allegheny Reservoir above Pittsburgh. During this same early period, the Ohio River Pollution Survey was completed through our joint efforts. This provided a very valuable background not only in the planning of Corps projects but to all other interests concerned with pollution of the Ohio River.

''Although water quality control was authorized as a specific purpose of only a few of our 214 completed reservoirs, some planned regulation for quality control is being made part of reservoir operating plans whenever it is consistent with the purposes for which the projects were authorized. Furthermore, in nearly all cases, the effect of regulation for authorized purposes is incidentally beneficial to water quality. Also, as we learn more about the effects of reservoir operations, steps can usually be taken to prevent adverse effects on water quality.

''On these completed projects two general courses of action are now possible. First, adjustments consistent with the authorized purposes of the projects, particularly in operations, can continue to be made in the light of experience and changing requirements for water quality. Second, completed projects can be restudied, particularly in conjunction with basin-wide studies for additional projects to meet future needs and, when feasible, recommendations made to Congress for modification of existing projects in the interests of quality control. We do not have authority to modify completed projects to include quality control without further action by Congress.

AUTHORIZED PROJECTS, UNDER CONSTRUCTION OR NOT STARTED

''Including those most recently authorized last year in Public Law 87-874, the Corps has authority to construct over 200 additional projects containing some 90 million acre-feet of storage. Of these, 81 reservoirs with 56 million acre-feet of capacity have been placed under construction. On all of the projects not yet started and on those where planning for certain phases of construction is still under way, the new Public Law 87-88 provides a basis for considering modifications of previously authorized plans to include provisions for flow regulation for water quality control if the following criteria are met:

''1. The modification of the project must be economically justified.

''2. The status of the planning and construction of the project must be such that it is practicable to accomplish the necessary changes without undue delay or unreasonable increase in cost over that which would have been incurred if water quality control had been originally authorized as a project purpose.

''3. The modification must not have a material adverse effect on the purposes for which the project was authorized originally.

''4. Satisfactory advice must be received from the Secretary of the Department of Health, Education, and Welfare relative to the need for and the value of the water quality control storage.

''5. The Congress must be informed of our intention to modify the project and of the

> views of the Secretary of the Department of Health, Education, and Welfare by letters to the Public Works and Appropriations Committees of the Senate and House.

> ''Last year we informed Congress that we were modifying two projects, John W. Flannagan Reservoir, Ohio River Basin, Virginia, and DeGray Reservoir, Ouachita River Basin, Arkansas, to include water quality control storage as a primary project purpose. We feel certain that many other authorized projects can be modified to include this important purpose as detailed planning studies proceed.''

Later in his paper Mr. Weber further stated:

> ''First, through training of current employees and in new hiring, we are equipping our own organization with personnel who encompass the wide range of disciplines needed to work out solutions to water problems. Our organization will include engineers who will try to solve water quality problems, and also biologists, economists, lawyers, and others needed to assimilate the complex considerations involved in modern water resource development plans.

> ''Secondly, we are trying to work out appropriate and effective arrangements for coordination and distribution of effort among all agencies who can and should contribute to solution of water problems. Insofar as water quality considerations are concerned, we are looking to the Department of Health, Education, and Welfare for both day-to-day advice and continuing, concurrent consideration of long-range studies.''

As mentioned in Public Law 660 as amended, consideration must be given to inclusion of storage for regulation of streamflow for the purposes of water quality control in impoundments constructed by the Bureau of Reclamation. In his talk, ''Relation of Regulation for Quality Control to Activities of the Bureau of Reclamation,''[143] given at the Symposium on Streamflow Regulation for Quality Control, April 3-5, 1963, sponsored by the Robert A. Taft Sanitary Engineering Center, Mr. D. R. Burnett, Chief, Division of Project Development, Bureau of Reclamation, discussed this point. By provision of various Acts, the Bureau of Reclamation is authorized to investigate, construct, and operate projects to provide water for irrigation and multiple purposes in the 17 contiguous Western States.

Two quotations summarized the position of the Bureau on this topic. Mr. Burnett noted:

> ''There are numerous specific project authorizations for construction and operation of Reclamation projects, some of which include works specifically for water quality control. ... In our work in the field of water quality, our experiences working with the Fish and Wildlife Service, the Public Health Service, and the National Park Service have been of paramount benefit. These agencies have contacts and sources of information not readily available to the Bureau. They also have biological knowledge outside the normal experience of our Reclamation personnel. The use of these sources by our planners has allowed us to progress in water quality control work at an accelerated pace.''

Other examples of current use of streamflow regulation for quality control are given in the following cases. In the Federal license to the Virginia Electric and Power Company, contained in the paper by Ragone and Peters,[10] is a requirement that states:

> ''In the event of temporary emergency conditions arising in performance of Requirements A and B, the Licensee will cooperate in good faith with the North Carolina State Department of Conservation and Development, North Carolina State Stream Sanitation Committee and/or North Carolina Wildlife Resources Commission to take such reasonable steps in regards to the reductions permitted by item (1) under Requirement B as may be proper to meet the emergency conditions and in the interest of maintaining during such emergency conditions water quality in keeping with the standards required of class ''C'' waters by order of the North Carolina State Stream Sanitation Committee effective September 1, 1957.''

This emergency provision was put into operation in 1962 when a fish kill due to presence of industrial wastes and low dissolved oxygen concentration conditions took place in the Roanoke River. During this time extra water was released from the Virginia Electric and Power Company Reservoirs for quality control.

Another example of streamflow regulation for quality control was in the Ohio River in the fall of 1963. Although the Corps of Engineers' reservoirs in the affected area did not specifically have additional capacity for quality control water, at the height of the drought the Corps happened to be lowering the level of their reservoirs from summer pool to winter pool elevation. During this time, they calculated they were supplying one-half of the flow in the Ohio River with these releases. Fortunately, the severe portion of the drought ended just as the level of the Corps of Engineers' reservoirs reached the winter pool elevation. Although there were some quality problems in the Ohio River at this time and a small fish kill, the situation probably would have been much worse had the Corps not been able to supply their water. Although this example of streamflow regulation for quality control was more of a lucky happenstance than a preplanned operation, it does show the potential advantages of streamflow regulation for quality control.

The preceding paragraphs indicate that, although all of the research needed in this area has not been done, projects designed to control water quality through streamflow regulation are a reality. Past research on the behavior of impounded water, together with accumulating experience in the operation of these projects, has brought us closer to an understanding of impoundments, thereby improving estimates of effects on water quality parameters and improving designs of projects. But, as this report has tried to indicate, in spite of the enormous literature on this subject, additional research work needs to be done.

Future Activities

In order to understand the behavior of impoundments and their influences on downstream water quality, additional research work is required. This work involves integrating existing information and filling in gaps, measuring effects in impoundments and streams that have been measured in other circumstances, and relating the environment to the measured effects. In some cases whole new areas of interest must be researched.

To begin to fill these needs, the Engineering Section at the Robert A. Taft Sanitary Engineering Center has started several laboratory, field, and desk-top studies on various phases of the overall problem. In each case the project began with an extensive literature review, for only through the integration of knowledge from a wide range of disciplines augmented by research can answers to the very complicated interrelated problems associated with impoundment behavior be forthcoming. The current (1964) state of the Engineering Section's research is summarized in an article by Symons, Weibel, and Robeck,[144] "Impoundment Influences on Water Quality," published in the *Journal of the American Water Works Association.*

Even when impoundment behavior is understood, other problems will still be associated with streamflow regulation projects. One question is: What criteria should be used to measure a good quality water? In the past, heavy emphasis has been placed upon dissolved oxygen concentrations, although even here the desirable level is subject to debate. Other quality characteristics must also be used to judge water, however. A second question concerns the monetary benefits associated with meeting these criteria. These must become available, as must the losses associated with temporary violation of one of these criteria. Currently, projects are often designed on a cost-minimization basis rather than on a benefit-maximization basis, because benefits are not truly calculable.

A third question involves the legal problems associated with streamflow regulation. Examples are the water rights problems and the problems caused by groups such as the recreationists who prevent water level changes that would interfere with their activities.

Finally, attention must be given to the methods other than streamflow regulation for the solution of water pollution problems. For example, if the construction and operating costs for water treatment plants were accurately known, the additional costs of treatment for the removal of a given pollutant could be compared with the cost of providing dilution water. Another possibility would be the use of one or more techniques being developed by the Advanced Waste Treatment Research Section at the Robert A. Taft Sanitary Engineering Center. In some instances, protection of water quality might be less expensive if advanced techniques were used, rather than streamflow regulation. A third example would be the possibility of aerating a stream mechanically, instead of supplying oxygen to a critical zone with impoundment releases. One million gallons of water with a dissolved oxygen concentration of 8 milligrams per liter contains only 66.5 pounds of oxygen; therefore, stream aeration may be less expensive than impoundment releases for providing oxygen. References 145 and 146 describe one of many such devices for aeration. Storage of wastes with controlled discharge is another useful technique for the protection of the oxygen resources of a stream. These alternatives are important and realistic, since streamflow regulation alone will not solve a basin-wide pollution problem. An example of the basin-wide approach to pollution abatement, which integrates numerous pollution control methods, is in the Ruhr Basin in Germany. This was discussed by Kneese[147] in an article, ''The Ruhr's Unique Approach to Water Management.''

Although these other problems exist, current research attention is focused on the impoundment itself. The research goals of this portion of the overall project are, in general, designed to measure the influences of impounding water on water quality, to relate these influences to the environment so that they may be projected to other environments, and to predict the influence of releases on the downstream water. Attainment of these goals will allow the prediction of the behavior and influence of new impoundments so that alternate sites, alternate methods of operation, or alternate treatment plans may be accurately integrated into an economic basin-wide analysis. Only through optimizing basin designs, using improved mathematical and economic techniques with better water quality behavior input data, and realistic alternative proposals can streamflow regulation projects for water quality control be considered truly successful. The information supplied by the research suggested in the body of this report should allow this hope to become a reality.

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PAPER NO. 2

IMPOUNDMENT INFLUENCES ON WATER QUALITY

J.M. Symons, S.R. Weibel, and G.G. Robeck

This paper reviews the necessity for research on water quality behavior and describes plans and early results in five research areas. These are: 1) the simulation of stratified reservoirs on a laboratory scale, 2) the rate and extent of the dissolution of iron and manganese from bottom muds in stratified reservoirs, 3) the dissolved oxygen budget in reservoirs, 4) the possibility of artificial destratification of reservoirs, and 5) the biologic transformations of nitrogen in reservoirs. Of these five areas, only number 2 was not carried to completion.

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INTRODUCTION

Two new areas in the field of water quality control hold great promise for the future. One is the use of impounded water for control and maintenance of downstream water quality. The other is the maintenance of good-quality water throughout an entire river basin by using systems analysis techniques to interrelate the various components that affect water quality in the river basin. Both techniques are in their infancy. The possibility of the major use of impounded water for streamflow regulation became a reality on July 20, 1961, when the amendment to Public Law

660[1] was enacted. Studies of the use of systems analysis for water quality management were begun some 6 years ago by the water resources group at Harvard University.[2]

Although both of these techniques have been used for water quality control, neither is being used to its fullest extent because of insufficient research. For example, difficulties are being experienced by mathematicians in preparing technologic functions that will express mathematically the behavior of individual components in a river basin, by economists in assessing monetary benefits or losses from the preservation or destruction of water quality in a river basin, and by engineers in accurately predicting changes that impoundments cause on water quality and the influence of impoundment releases on downstream water quality.

In response to the passage of the amendment to Public Law 660, the engineering section at the Robert A. Taft Sanitary Engineering Center undertook research into the influences of impoundments on the behavior of water quality.

To paraphrase the portions of Public Law 660, as amended, that relate to these projects, Section 4(d)(1) states that methods of evaluating the effects on water quality of regulated streamflow must be developed through research. Section 2(b)(1-2) states that federal agencies constructing impoundments must give consideration, but not necessarily approval, to the inclusion of storage capacity for the purpose of water quality control. Further, the need for and the value of this storage capacity shall be determined by the agency, either positively or negatively, with the advice of the Secretary of the Department of Health, Education, and Welfare.

This large research mandate has been divided initially into five parts: (1) characterization and treatment of storm water runoff from urban areas, (2) characterization and treatment of storm water runoff from rural areas, (3) investigation and re-evaluation of stream pollution criteria, (4) determination and prediction of the behavior of water quality in flowing streams, and (5) determination, and prediction of the behavior of water quality in impoundments. Part 1 has recently been discussed by Weibel, Anderson, and Woodward,[3] and Part 2 has been reported by Weidner, et al.[4] Part 5 is discussed below.

General Problem

The environmental conditions in an impoundment are different from those in a flowing stream in many ways, including change in water depth, increased detention time, and the possibility of thermal stratification. Each has an influence upon the normal biologic, physical, and chemical processes that cause changes in water quality.

Many of these processes have been studied under flowing-stream conditions but not in impoundment environments. Here the additional detention time allows natural processes to proceed far beyond the extent possible in a flowing stream. In addition, if thermal stratification exists in an impoundment, the bottom portion of the water is trapped and does not contact the atmosphere or aerated water. This condition also has a marked effect on the natural processes that occur in the water.

The purpose of the pertinent sections of Public Law 660, as amended, was to establish streamflow regulation systems that would provide for impounding water during times of high streamflow and releasing the water at times of low natural streamflow. Such releases would supply additional water downstream and would serve to alleviate any downstream water quality problems. If such a system is to be successful, the quality of impoundment releases must be predictable and managed to ensure that the downstream influence is beneficial, not adverse. At present, the influences of an impoundment environment on water quality changes are uncertain. If water were impounded, one could not, therefore, state with certainty what the quality of this water would be if it were released at some time in the future for the purpose of streamflow regulation. Of course, predictions of impoundment and stream behavior are being made despite the need for greater knowledge.

This inability may be somewhat surprising, because in considering the literature of stream and impoundment behavior one is immediately impressed by the vast number of writings on the subject. There are at least two reasons for this uncertainty, despite the extent of research. First, much of the data, though generally applicable, were not taken with impoundment behavior specifically in the researcher's mind. For example, extensive research by limnologists and aquatic biologists has related the growth of algae to light intensity and penetration. Little attention has been paid, however, to the increased penetration that results from impounding a turbid river water and to the potential organism growth problem that this clarification may pose.

Second, many of the investigations have not been fundamental enough to allow projection of the phenomena observed into a new set of environmental conditions. For example, thermal stratification has been noted in impoundments for years, yet the necessary set of environmental circumstances that enhance the possibility of thermal stratification has never been completely delineated.

It is therefore necessary to investigate new relationships thoroughly and to relate previously observed results to the environmental factors that caused them, so that in a different environment different results could be predicted. These are the two overriding research needs in the entire area of impoundment behavior; until they are met, streamflow regulation projects for quality control will not be founded on a firm scientific basis.

Research Approach

There is a problem of determining the better approach to water quality-forecasting techniques. One method would measure the changes of a particular water quality parameter in several different natural environments. Then, by relating the rate and extent of the measured change to the total environment in which this change took place, an all-inclusive predictive scheme relating environment to parameter change could be developed. In a new situation one would need only to measure the essential environmental factors, consult the developed theory, and the change in the particular parameter could be predicted for the new environment. The development of such a scheme would be ideal in that the particular quality parameter would be henceforth completely predictable. The only difficulty with this approach is that the development of truly all-inclusive theories may be beyond the scope of current ability, as so many factors in a natural environment exert subtle influences on the change in water quality. Currently, however, the extent of this difficulty is not known.

An alternative approach would be the development of a series of operational tests that would allow the measurement of the behavior of a water quality parameter under a controlled environment sufficiently similar to the natural one that the data would be applicable to the field situation. This is what the 5-day BOD test tries to do for the oxygen budget of a flowing stream, although the bottle environment is sufficiently different from the natural one that the data are somewhat inapplicable to most field situations. Although this approach would require several tests for each new impoundment, it might be a meaningful one if a relatively simple set of operational tests could be developed.

At this time, both avenues are being explored. In current research the environment is being sufficiently controlled to permit correlation of observed results with pertinent environmental factors. Simultaneously, the testing procedures are being designed in such a manner that if a general relationship between the environment and results cannot be developed, the operational tests are sufficiently realistic that the data taken are applicable to a field situation. At the conclusion of these studies it is hoped, therefore, that either useful theories or useful operational tests, or both, will have been developed.

Background Information

As the literature in the area of streamflow regulation for water quality control, with emphasis on the behavior of impoundments as they influence water quality, is so large, it cannot be discussed within the scope of this article. A comprehensive review, in which over 600 references are reduced to the 147 that seem most pertinent, has been prepared and issued as an Environmental Health Series report.[5] A general view of present knowledge and thinking in this area may be obtained by reviewing References 6-19.

In spite of the previous work in this area, streamflow regulation cannot be successful without further research. Major research gaps exist in the following areas: (1) improvement of techniques in the field of systems analysis, (2) behavior of such parameters as iron, manganese, nitrogen, phosphorus, and oxygen in impoundment environments, (3) influence of continuing fertilization on water quality in impoundments, (4) influence of the physical environment on impoundment behavior, and (5) effect of impoundment releases on downstream water quality. These are not the only areas in which additional research is needed, but they are some of the more important ones.

Initial Projects

From the large number of possible research projects, the engineering section at the Robert A. Taft Sanitary Engineering Center chose the five below for initial investigation.

Simulation of impoundments on a small scale. To permit fundamental studies on the behavior of water quality in impoundments, an apparatus to make possible measurements under controlled conditions must be available. Although any data taken on a small-scale apparatus would eventually have to be checked under field conditions, the natural environment is so uncontrolled that fundamental studies are difficult.

Dissolution of iron and manganese. The dissolution of iron and manganese in the bottom waters of stratified impoundments and the subsequent release of water containing relatively high concentrations of these materials causes a deterioration of downstream water quality. This is particularly true if the downstream water is used for municipal drinking water or industrial process water. It is therefore important to understand not only the mechanism by which iron and manganese is dissolved, but also the factors that control the rate and extent of this dissolution.

Studies on the dissolved oxygen budget. Traditionally the dissolved oxygen (DO) budget has been the most important factor in stream pollution studies. Although this is no longer the only water quality parameter of interest, it is still extremely important. Understanding the influence of an impoundment environment on the DO budget is, therefore, essential, as the DO concentration influences the behavior of so many other water quality parameters.

Artificial destratification of impoundments. There is general agreement that stratification of impoundments leads to creation of poor-quality water in the hypolimnion. If sufficient energy could be added to an impoundment to overcome the forces of stratification, this deterioration of the bottom waters could be prevented. The potential benefits are great enough to justify the investigation of mechanical destratification.

Biologic transformation of nitrogen. Although the nitrogen concentration in most impoundments is relatively low, the changes from one nitrogen form to another caused by the biologic populations in impoundments are important. Oxidation of ammonia consumes dissolved oxygen, reduction of oxidized nitrogen supplies oxygen, and nitrogen plays a vital role in the synthesis of new organisms, both bacteria and plankton. These transformations become, therefore, important to investigate.

LITERATURE REVIEW, THEORETIC CONSIDERATIONS, AND EXPERIMENTAL APPROACH

Only two of these projects are sufficiently advanced to allow the presentation of some preliminary data. This article, however, serves as an announcement of the initiation of these studies, as a review of the literature and theoretic considerations pertinent to these research projects, and as a discussion of the experimental approaches being taken. As these projects advance and are completed, future articles will discuss them in detail.

Small-Scale Simulation

The literature contains few descriptions of attempts to simulate impoundment conditions on a small scale. Such studies, however, seem important, as any fundamental research on impoundment behavior should be conducted under controlled conditions. Two attempts are being made to develop laboratory equipment that permits such studies. One is a 3-foot-diameter, 12-foot-high, 500-gallon tank in which thermal stratification is created. The other is a series of 6-inch-diameter, 5.2-foot columns very similar to those described by Sylvester and Carlson.[20]

Dissolution of Iron and Manganese

Relatively high concentrations of iron and manganese in impoundment releases can cause serious trouble downstream. The ability to predict under what circumstances and to what extent iron and manganese will go into solution in a given environment is, therefore, important. This general area has been studied by Ingols and Wilroy,[21] Wilroy and Ingols,[22] Stumm and Lee,[23] and Morgan and Stumm.[24] The iron cycle is summarized in Figure 1. The presence of dissolved oxygen in the epilimnion of a stratified impoundment and the absence of oxygen in the hypolimnion play a critical role in the overall iron cycle.

A summary of the manganese cycle according to Ingols and Wilroy[21] is given in Figure 2. Morgan and Stumm[24] have a different concept of the manganese cycle, and a summary of their ideas is shown in Figure 3.

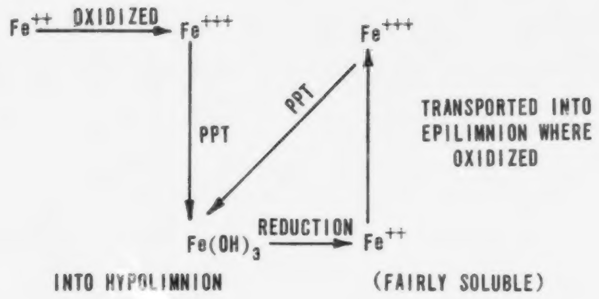
Fe++ OXIDIZED Fe+++
Fe+++
PPT
PPT
TRANSPORTED INTO EPILIMNION WHERE OXIDIZED
Fe(OH)3 REDUCTION Fe++
INTO HYPOLIMNION
(FAIRLY SOLUBLE)

Figure 1. Summary of iron cycle, after Stumm, et al.[23,24]

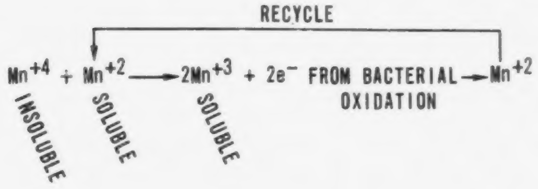
RECYCLE
Mn+4 + Mn+2 → 2Mn+3 + 2e⁻ FROM BACTERIAL OXIDATION → Mn+2
INSOLUBLE
SOLUBLE
SOLUBLE

Figure 2. Summary of manganese cycle, after Ingols and Wilroy.[21]

Ingols and Wilroy[21] suggest that bacterial oxidation is intimately connected with the manganese cycle, and that the manganese may be intracellularly transformed. Morgan and Stumm[24] suggest that the extracellular results of biologic activity–such as change in pH, removal of oxygen in the bottom waters, or both–influence the manganese cycle.

As a first phase of the overall study, an attempt will be made to relate the rate and extent of the dissolution of iron and manganese to the original iron and manganese content of the bottom muds of impoundments. This relationship, at least with respect to manganese, has been suggested by Wilroy and Ingols.[22]

Dissolved Oxygen Budget

The DO concentration in impounded waters is one important water quality parameter that is influenced by many

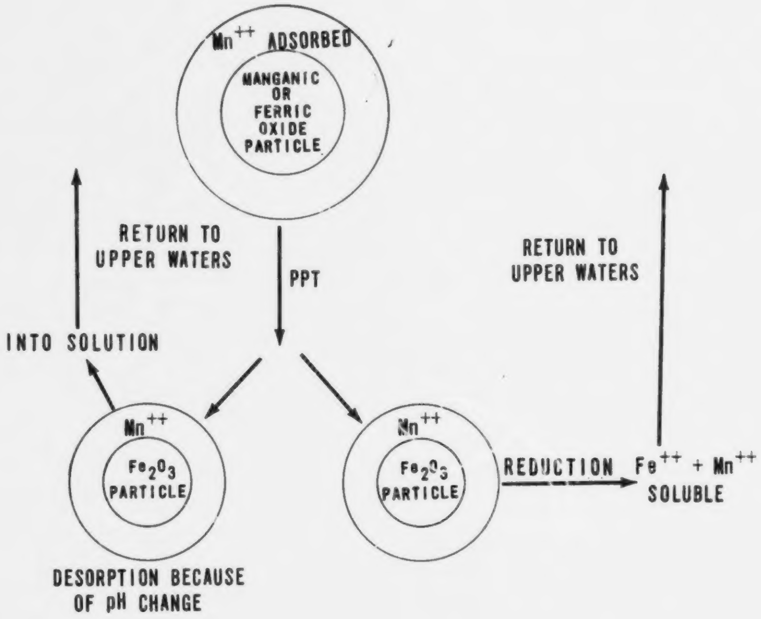
Mn++ ADSORBED
MANGANIC OR FERRIC OXIDE PARTICLE
RETURN TO UPPER WATERS
PPT
RETURN TO UPPER WATERS
INTO SOLUTION
Mn++
Fe2O3 PARTICLE
Mn++
Fe2O3 PARTICLE
REDUCTION
Fe++ + Mn++ SOLUBLE
DESORPTION BECAUSE OF pH CHANGE

Figure 3. Summary of manganese cycle, after Morgan and Stumm.[24]

factors. Oxygen is removed from an impoundment through the metabolism or respiration or both of the plants and animals contained in the water, as well as through the loss of oxygen to the atmosphere if the surface waters become supersaturated with oxygen. Oxygen is added to an impoundment through the metabolism of plants and dissolution from the overlying atmosphere. Inflow and outflow of oxygen with the water passing through the impoundment is neglected in this analysis. Considerable attention has been given to atmospheric reaeration, for example, by Churchill, Elmore, and Buckingham,[25] and Krenkel and Orlob,[26] but these and similar studies have more application to flowing stream waters than to very slowly moving impoundment waters. Photosynthetic oxygen has been less extensively studied, except for the work by Hull.[27-30] Hull has almost exclusively used the ''light-'' and ''dark-'' bottle technique for evaluating photosynthetic oxygenation, although this technique has been criticized by Pratt and Berkson,[31] Verduin,[32] Kaplovsky,[33] and Strickland,[34] and by Hull himself. Hull has used the method, with reservations concerning its accuracy, to show the general significance of photosynthesis in the oxygen balance of polluted waters.

Study in this area seeks to refine and extend the measurements made by Hull by using a similar technique, but one that avoids some of the disadvantages of the traditional light- and dark-bottle method. In addition, measurements are made that allow an evaluation of the reaeration coefficient in impoundments. This study makes use of three DO probes* that have been especially modified to permit the detection of very small changes in DO concentration. One probe is placed in a 14-liter container made of clear plastic to permit the admission of sunlight. A second probe is placed in a 14-liter container made of black plastic to prevent the admission of sunlight. The third probe is external to the two containers.

The containers are designed with spring-loaded trapdoors at the top and bottom so they can be lowered, open, to a given water depth, the trapdoors closed by tripping a trigger with a nylon line, and a 14-liter volume of water from that depth trapped inside the containers. The DO concentration in these two containers as well as in the external water at the same depth is then monitored over a given period of time to detect the changes. A photograph of the light container and probe in both the open and closed positions is shown in Figure 4. The triggers are on each side with the messenger lines attached.

During the daylight hours the DO concentration in the light box should be influenced by the respiration and metabolism of the microorganisms in the water at that depth as well as by the oxygen production of the algae at the same depth. The oxygen concentration in the dark box should be influenced only by the respiration of the microorganisms in the water and their metabolism of organic matter. The oxygen concentration determined by the external probe should be influenced not only by the metabolism and respiration of the microorganisms and the oxygen production of the algae, but also by atmospheric reaeration. Theoretically the dark box should measure the oxygen demand of the water, the difference between the oxygen concentration in the dark and light box should be a measure of photosynthetic oxygen production, and the

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*Model SM122S; a product of Pro-Tech, Inc., Broomall, Pa. [Similar DO probes currently (1969) manufactured by Automated Environmental Systems, Inc., Woodbury, Long Island, New York].

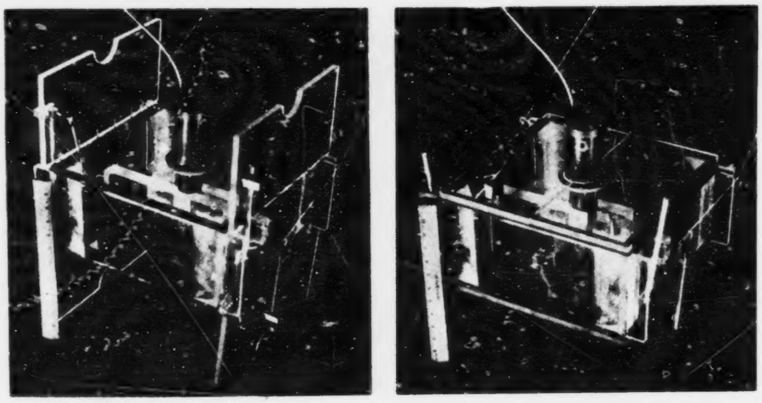

Figure 4. ''Light'' container and DO probe in open and closed position - note 12-inch scale in left foreground.

difference in oxygen concentration between the light box and the external probe should be a measure of the influence of atmospheric reaeration.

Even though Verduin[32] feels that any time a sample of water is placed in a container the environment is altered sufficiently so that the results are somewhat atypical, the apparatus described above should avoid some of the chief criticisms of traditional light- and dark-bottle experiments. Verduin[32] notes that there is no turbulence in traditional light- and dark-bottle experiments, but in the apparatus described above, a self-contained magnetic mixer keeps the liquid inside the containers continuously stirred. This facilitates the flow of food to, and of waste products away from the organisms contained inside the boxes.

In addition, as this apparatus traps a relatively undisturbed sample of water at its original water depth–rather than bringing water from a given depth to the surface, filling bottles, and returning them to the original depth–the objections of Vaccaro and Ryther[35] relating to the bactericidal effect of sunlight on plankton growing in dim light are avoided, although this may also be avoided by filling the bottles between sunset and sunrise. Finally, by continuously monitoring the DO changes under each of the three conditions, the rates of change should be determinable over short periods under relatively constant conditions of temperature, wind direction, wind intensity, sunlight, and so forth. This should allow correlations to be made between environment and dissolved oxygen budget.

Artificial Destratification

The general deterioration of water quality in the hypolimnion of a stratified impoundment has been recognized by many investigators, including Kittrell[16,36] and Ingols.[17,18] If adding sufficient mechanical energy to overcome the forces of stratification caused by the difference of density of water at differential temperatures would be possible, no anaerobic zone of water would develop, and the overall quality of water in the impoundment should be better. There may, however, be some hazard in this practice, because the bringing of settled phosphorus to the surface may cause an increased growth of algae.

Since 1952, when the first reported attempt at artificial destratification was made by Hooper, Ball, and Tanner,[37] other attempts to mix impoundments have been reported. All the experiments except one were successful, and investigators reported the presence of oxygen in the lower stratum of water. Hooper, Ball, and Tanner[37] used a centrifugal pump for circulation, but an air piston device for accomplishing the same purpose was later described by Bryan.[38] All attempts to date (1964) have been made on relatively small bodies of water, 40,000 acre-feet maximum, so that the practicality of mixing a large storage impoundment has yet to be demonstrated.

In a stratified impoundment, the thermocline is defined as the zone of water in which the temperature gradient is equal to or greater than a decline of 1°C per meter of depth. The generalized theory is that the water above the thermocline, the epilimnion, is mixed by the wind, while the water below the thermocline, the hypolimnion, is stagnant.

A classical thermocline does not have to exist to have stratification, however. As shown in Figure 5, the decrease in density for each degree Centigrade is much greater at warmer temperatures than at colder ones. For example, the difference in density between 25°C and 30°C water (1.40 mg/ml) is 2.34 times greater than the difference in density between 10°C and 15°C water (0.60

mg/ml). This means that even if an impoundment showed a rather modest temperature gradient never reaching the intensity of a thermocline, it could still be density stratified. It is, therefore, the density profile, not the temperature profile, that determines the thickness of the stratum of water that must be mechanically mixed if artificial circulation is to be accomplished.

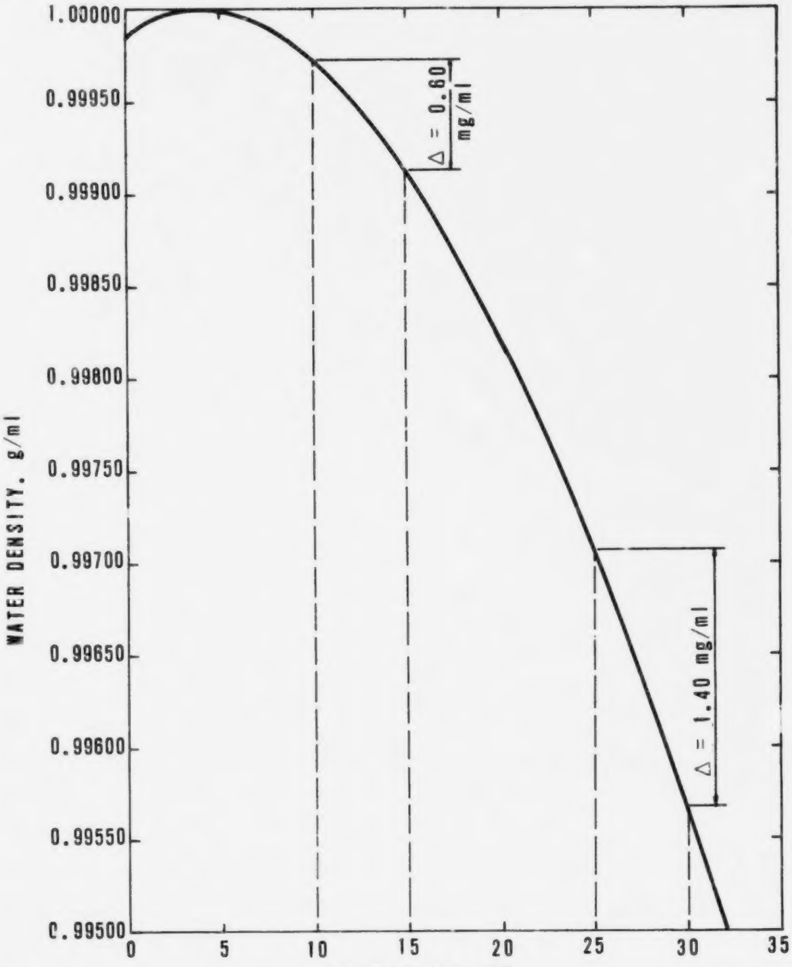

Figure 5. Water densities for various temperatures.

As the volume-depth curve of most impoundments is not linear–that is, a rather small volume of water relative to the overall volume of the impoundment will be contained in the lower stratum that must be raised to the surface–a relatively small volume of water may actually have to be moved. A typical pattern for a small lake in a vicinity of Cincinnati is shown in Figure 6. Here only 20 percent of the volume lies below the mid-depth.

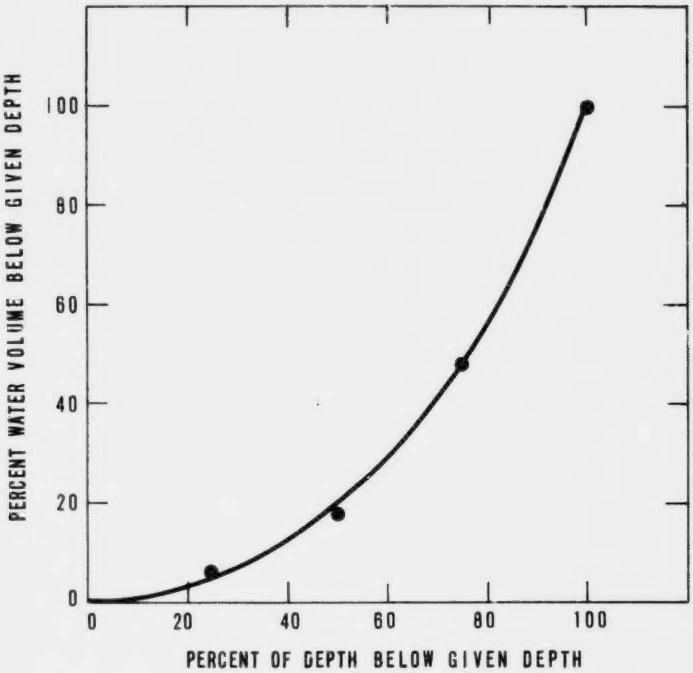

Figure 6. Volume vs. depth curve for Stonelick Lake, Ohio.

ENERGY REQUIREMENTS

In addition to this favorable feature, the total energy that must be applied to the water is comparatively small, because the difference in elevation head that must be overcome to raise the relatively high density water at the bottom to the surface is very small. For another small lake near Cincinnati that has a summer surface temperature of 25°C and a bottom temperature of about 11°C, and is 40 feet deep at the dam, the actual difference in elevation head through which the dense water has to be raised to bring it to the surface is only 0.05 foot. This is illustrated in Figure 7. If the suction line of the pump were filled with bottom water, it would stand to a depth of 39.95 feet to balance the weight of an external column of water with the same cross-section but with variable density. The difference is, therefore, 0.05 foot.

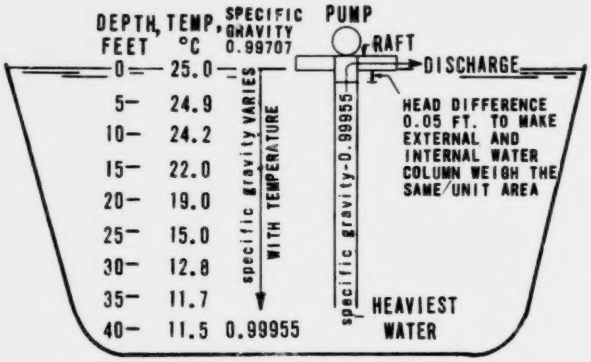
DEPTH, FEET
TEMP, °C
SPECIFIC GRAVITY 0.99707
PUMP
RAFT
DISCHARGE
0— 25.0
5— 24.9
10— 24.2
15— 22.0
20— 19.0
25— 15.0
30— 12.8
35— 11.7
40— 11.5
0.99955
specific gravity VARIES WITH TEMPERATURE
specific gravity-0.99955
HEAD DIFFERENCE 0.05 FT. TO MAKE EXTERNAL AND INTERNAL WATER COLUMN WEIGH THE SAME/UNIT AREA
HEAVIEST WATER

Figure 7. Theoretical elevation head to lift bottom water to surface-Lake Cowan, Ohio.

Except for losses, the energy required to lift the bottom 5,000 acre-feet of this 12,000-acre-feet lake to the surface would be only 343 horsepower-hours. This calculation neglects any energy input of the wind, which might be significant in some cases. Of course, much more than 343 horsepower-hours of energy would have to be supplied to overcome the losses inherent in such a pumping system, but the basic energy requirements are very small. Similar calculations have been made by Kittrell.[16]

POWER REQUIREMENTS

In addition to the consideration of the work requirements for mixing, the power–the rate of doing work–must also be considered. In the example above, 1 horsepower applied to the water for 343 hours (slightly more than 14 days) would do the necessary work. As

$$\text{Power} = Q\gamma H, \qquad (1)$$

the pumping rate would have to be 176 cubic feet per second (78,500 gpm) for a head of 0.05 foot to apply 1 horsepower of power to the water. This pumping rate would theoretically mix the lake every 2 weeks. Consequently, even though the total energy is not large, the power requirements might be large.

The power requirements could be reduced if the energy could be applied over a longer time. This would, however, mix the water more slowly. The rate at which an impoundment must be mixed to prevent anaerobic conditions developing in the deeper waters is controlled by the rate of oxygen demand that would exist in these deeper waters when they are isolated from the atmosphere. This rate of oxygen demand therefore controls the power input necessary in any given impoundment under consideration.

To determine the feasibility of artificial mixing of a stratified impoundment, experiments are under way to mix a small lake similar to that described in Figure 6. The mixing device is a 2,880-gallons per minute (13 acre-feet per day) mixed-flow-type pump driven by a 16-horsepower gasoline engine. The pump will be mounted on a raft positioned over the deepest part of the lake and will draw water from just above the bottom and discharge it at the surface. Water temperature and water quality analyses will be made to determine the efficiency of destratification and its influence on water quality. In addition, attempts will be made to determine whether or not all the water to be mixed must be passed through the pump, or whether a slow inertial current will be generated in the impoundment that will entrain additional bottom water and thereby reduce the amount of water that must actually be passed through the pump. If this study is successful, the destratification of larger bodies of water with larger pieces of equipment will be attempted in the future.

Nitrogen Transformations

The biologic transformations of nitrogen have been studied by many investigators through the years and are effectively summarized by Hutchinson.[13] The overall cycle is summarized in Figure 8. If the future water quality in an impoundment is to be truly predictable, all the potential changes in nitrogen forms must be understood.

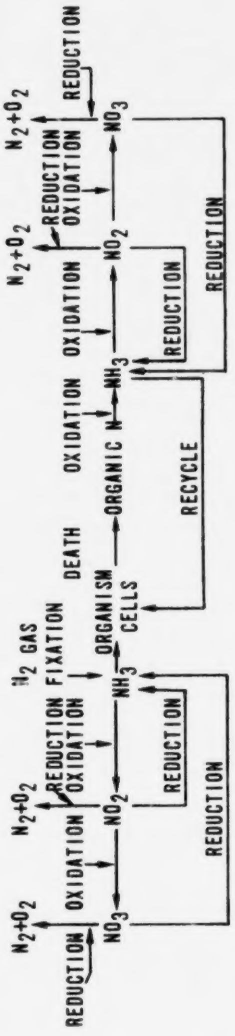
REDUCTION
N_2+O_2
NO_3
OXIDATION
N_2+O_2
REDUCTION
OXIDATION
NO_2
REDUCTION
REDUCTION
NH_3
N_2 GAS
FIXATION
ORGANISM CELLS
DEATH
ORGANIC N
OXIDATION
RECYCLE
NH_3
OXIDATION
REDUCTION
NO_2
N_2+O_2
REDUCTION
OXIDATION
REDUCTION
NO_3
N_2+O_2
REDUCTION

Figure 8. Summary of nitrogen cycle, after Hutchinson.[13]

One of the principal concerns in previous nitrogen transformation investigations has been the inability to account for all the nitrogen as it changes form, that is, the inability to make a nitrogen balance. This inability has been noticed by Symons[39] in activated-sludge studies, by Robeck and his co-workers in unpublished results taken from soil lysimeter studies, and by Leclerc[40] in studies made in streams. Theoretically, under completely aerobic conditions, the analyses for ammonia, nitrite, nitrate, and organic nitrogen concentrations should be sufficient to make a nitrogen balance. As the previous three studies were all made under generally aerobic conditions, the inability to make a nitrogen balance resulted either from microanaerobic environments in the system that caused some reduction of the oxidized nitrogen forms and associated loss in nitrogen gas, or from errors in the analyses of the nitrogen forms. If the latter was the case, the fault most probably was with the nitrate determination as the phenoldisulfonic acid method for nitrate analyses is somewhat unreliable.

Investigations in this area have two purposes: (1) to show whether or not a nitrogen balance can be made under controlled, completely aerobic conditions, and (2) to study the influence of the environment on the biologic transformations of nitrogen, if a balance can be made.

CURRENT RESEARCH: EQUIPMENT, METHODS, AND RESULTS

The two projects that have progressed sufficiently to permit the presentation of some preliminary data are discussed below.

Small-Scale Simulation

Equipment and methods. Figure 9 shows a cross section of the 500-gallon tank that has been developed to simulate a stratified impoundment. By cooling the lower portion of the water column, heating the upper portion, and insulating the sides of the metal tank, both thermal and chemical stratification have been accomplished. Upper-stratum circulation and reaeration is accomplished by blowing water-saturated air (water saturated to reduce evaporation) over the surface of the liquid for 12 hours a day, and by continually stirring the top of the tank

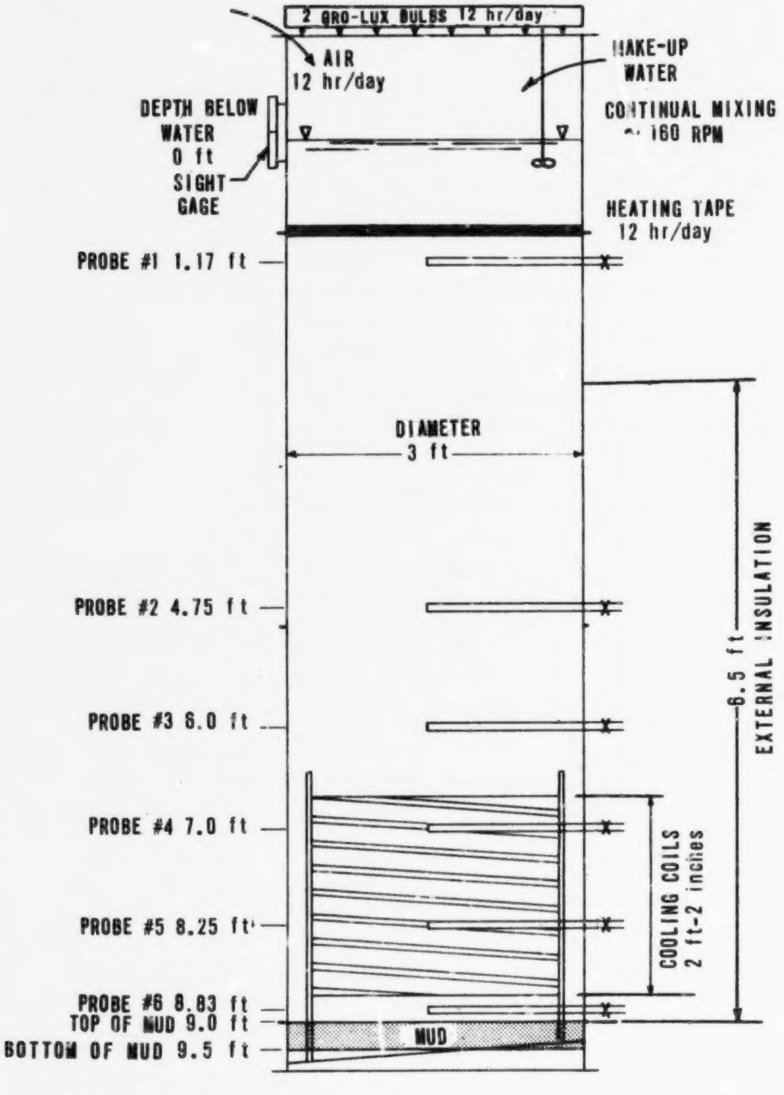
2 GRO-LUX BULBS 12 hr/day
AIR
12 hr/day
MAKE-UP
WATER
DEPTH BELOW
WATER
0 ft
CONTINUAL MIXING
~ 160 RPM
SIGHT
GAGE
HEATING TAPE
12 hr/day
PROBE #1 1.17 ft
DIAMETER
3 ft
6.5 ft
EXTERNAL INSULATION
PROBE #2 4.75 ft
PROBE #3 6.0 ft
PROBE #4 7.0 ft
COOLING COILS
2 ft-2 inches
PROBE #5 8.25 ft
PROBE #6 8.83 ft
TOP OF MUD 9.0 ft
MUD
BOTTOM OF MUD 9.5 ft

Figure 9. Stratified water column, 500-gallon capacity.

with a small laboratory mixer. Algal growth is stimulated by 12-hours-per-day illumination with fluorescent light bulbs.* Six sampling ports allow withdrawal of a sample from several levels.

This apparatus is successful but large and bulky. An attempt has, therefore, been made to accomplish similar simulation on a smaller scale with a soil-water contact column 6 inches in diameter and 5.2 feet long. About 4 inches of soil is placed in the bottom of this column, and overlaying water is added. This is a static apparatus, and it is hoped that upon standing, chemical (but probably not thermal) stratification will occur. A cross section of this column is shown in Figure 10.

Results. Figures 11 to 13 show the extent of thermal and chemical stratification in the large simulated impoundment. Figure 11 shows a very sharp temperature gradient between the 6- and 7-foot depths. The other data show that the unit is chemically stratified, that the DO concentration is practically zero in the hypolimnion, and that the epilimnion has a DO concentration nearly 90 percent of saturation.

The data in Figures 12 and 13 do not show as pronounced a level of chemical stratification as the DO, iron, and manganese data shown in Figure 11. Figure 12 does, however, show that the alkalinity is much higher in the cold bottom waters, and the nitrogen data in Figure 13 show that the ammonia nitrogen concentration in the bottom waters is also higher than in the top waters. This may be because of ammonia alkalinity, which is produced during the anaerobic degradation of proteinaceous material that may have originally been in the bottom muds or settled into the bottom waters. Figure 13 also shows that there is an absence of oxidized nitrogen forms in the anaerobic bottom waters, which is to be expected. The discontinuity in the ΣN curve, which occurs at the 6-foot level, is probably caused by analytic error.

After 220 days of operation, a sample of water was collected from the top probe, which is about 1 foot below the surface of the water. This sample was examined microscopically to determine the number of free-floating algae in the surface water. A relatively low total count of 498 per millimeter was obtained.

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*Gro-lux; a product of Sylvania Electric Products, Inc., New York, N.Y.

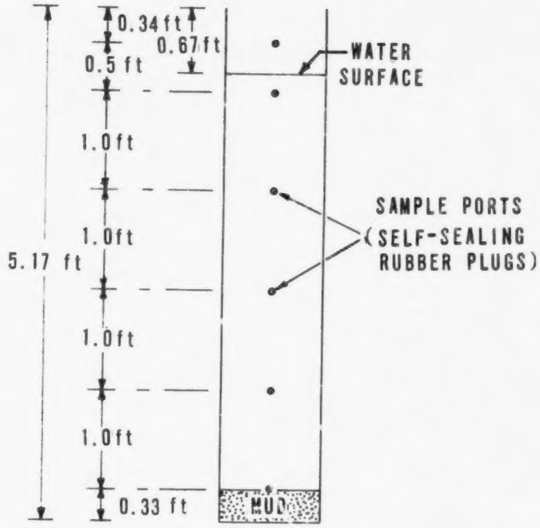

Figure 10. 6-inch soil - water contact column, after Sylvester and Carlson.[17]

Before the study, the apparatus was filled with tap water and allowed to stratify thermally. The rate of mixing of the surface waters was then determined. The introduction of a relatively large quantity of nitrite nitrogen into the surface waters, followed by periodic analyses of the nitrite nitrogen concentration at various levels in the tank, indicated that the top water was mixed every 20 minutes and that no nitrite nitrogen penetrated temperature gradient after 12 days of analyses. This finding indicates that the apparatus behaves in a manner similar to that of a stratified impoundment in the field, mixing within the surface water, but with little exchange between the top and bottom water.

The initial performance of the 6-inch column is shown in Figure 14. Figure 14 shows the decline of the DO concentration with time in the liquid layer just over the mud during the first 23 days of operation. In this experiment, Cincinnati tap water was contacted with soil containing a relatively high percentage of organic matter. Figure 14 also shows a vertical profile of DO concentration taken on the 23rd day, which indicates that some chemical stratification was occurring. A similar apparatus, in which the

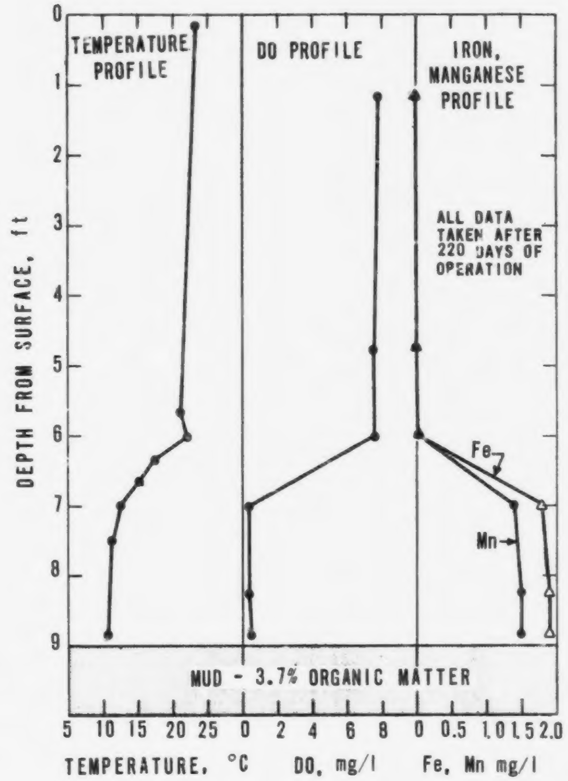

Figure 11. Data from simulated stratified impoundment.

water contacted a sandy bottom containing a relatively low percentage of organic matter, showed a much slower decline in the oxygen concentration in the water just above the bottom material. Although initially successful, subsequent data indicated mixing was occurring. This was confirmed by using strontium-85 as a tracer. The apparatus will be modified by adding heating tapes around the upper portions to ensure stratification. Pertinent analytic techniques have been miniaturized to permit the determination of DO, alkalinity, iron, and manganese concentrations, and pH with only 30 milliliters of sample.

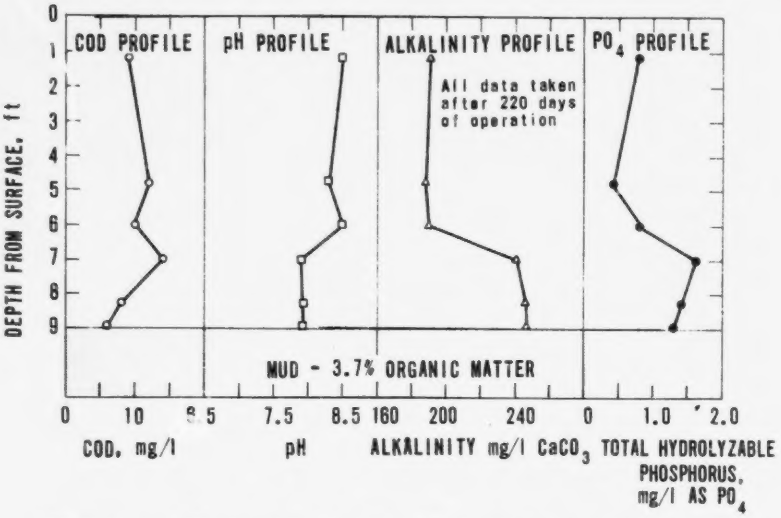

Figure 12. Data from simulated stratified impoundment.

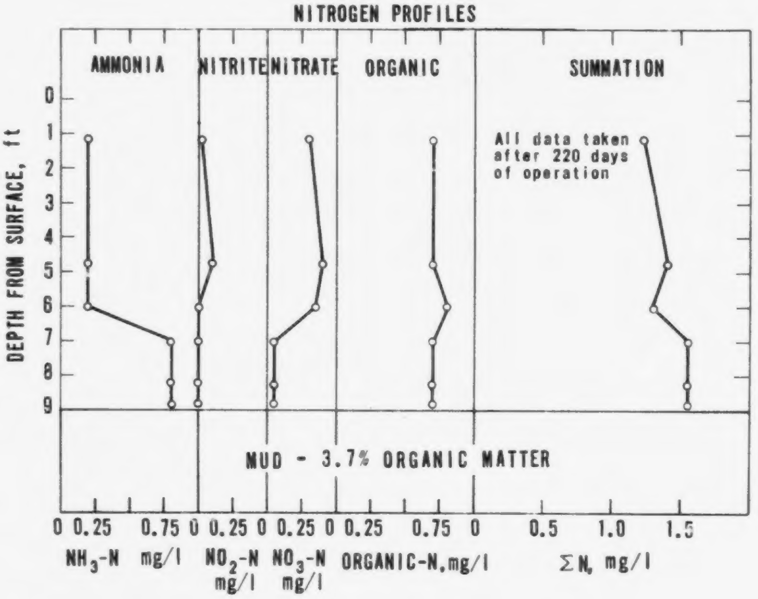

Figure 13. Data from simulated stratified impoundment.

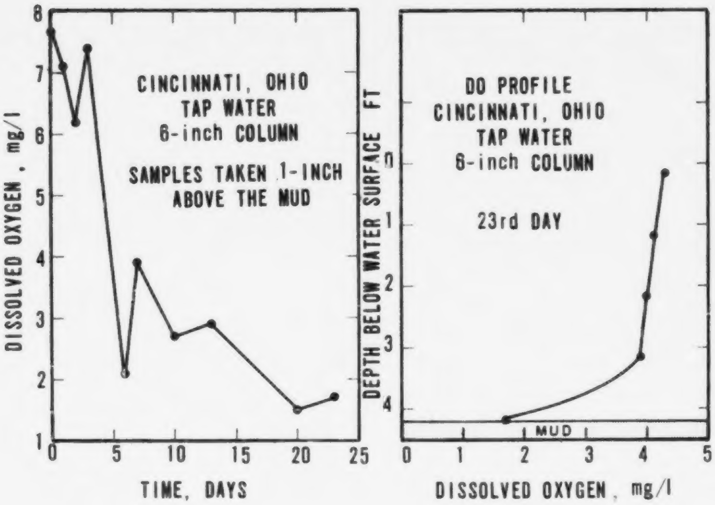

Figure 14. Performance of 6-inch soil - water contact column.

Nitrogen Transformations

Equipment and methods. The experimental method used was a modification of the one reported by Konecky, Kelly, Symons, and McCarty.[41] The apparatus consisted of several 5-gallon glass jars that had provisions for mechanical mixing through the top, introduction of air to maintain aerobic conditions, and a sampling siphon. The basic test medium was fortified distilled water. The distilled water was fortified by a yeast nitrogen base,* which supplied a wide variety of vitamins, minerals, and amino acids, as well as ammonia nitrogen in the form of ammonium sulfate. When 31.5 milligrams of this powder per liter was used, the resulting ammonia nitrogen concentration was 5 milligrams per liter. When more than 5 milligrams of ammonia nitrogen per liter was required for experimentation, the excess was added as ammonium sulfate. The system was buffered with sodium bicarbonate and monopotassium dihydrogen phosphate. The final pH of the solution was 7.5, and the final alkalinity was about 200 milligrams per liter as $CaCO_3$. This was a good, clean, synthetic river water that was nutritionally balanced to permit rapid growth of autotrophic bacteria. In addition, the medium was free of substances that would interfere with any of the subsequent analyses.

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*A product of Difco Labs., Inc., Detroit, Mich.

The seed for these experiments was obtained from an operating bench scale activated-sludge plant that was actively nitrifying. Mixed liquor was washed twice in distilled water to remove soluble carryover products from the activated-sludge system, and the volatile suspended-solids concentration was determined. The solids were then blended to disperse the floc, and a portion was added to the synthetic river water to supply the desired volatile suspended-solids concentration at the beginning of the test. Although the seed was not standardized, it was washed and was added in relatively low concentrations. This avoided the addition of unnecessary extraneous matter and approximated the volatile suspended-solids concentration that might be found in streams and impoundments.

Periodically during a test, samples were withdrawn and analyzed for pH, and concentrations of alkalinity, chemical oxygen demand, DO, and all four nitrogen forms. In addition, samples were withdrawn at similar intervals, and the ammonia-oxidizing and nitrite-oxidizing microorganism populations were determined.

The analytic techniques used to determine the nitrogen concentrations and nitrifying populations were unique and are discussed in some detail. All the nitrogen forms were determined either wholly or in part through the use of an automatic analyzer.* The basic reagents used in the individual analyses are given below. In each case the samples were softened by pH adjustment and filtered to remove hardness precipitate and turbidity before colorimetric reactions were begun.

Ammonia nitrogen: Ammonia nitrogen concentrations were determined by reacting the samples with sodium hypochlorite and sodium phenolate.

Nitrite nitrogen: The nitrite nitrogen concentrations were determined by reacting the samples with sulfanilamide and N(1-naphthyl)-ethylenediamine dihydrochloride.

Nitrate nitrogen: The nitrate nitrogen concentrations were determined on the samples by reducing them with alkaline hydrazine, and following this reduction, by

*Technicon AutoAnalyzer; a product of Technicon Corporation, Tarrytown, New York.

diazotization using the reagents noted in the nitrite nitrogen section above.

Organic nitrogen: The organic nitrogen was determined by use of the method given in *Standard Methods*[42] with two exceptions: (1) the procedure was done on a relatively small sample, that is, on a micro scale, and (2) the ammonia nitrogen distilled after digestion was collected in a weak solution of sulfuric acid. After collection, the ammonia nitrogen concentration was determined by the procedure given under the ammonia nitrogen section above.

Nitrifying populations: The ammonia- and nitrite-oxidizing microorganism populations were determined at various times during the experiments by use of an MPN (5-tube replicate serial dilution) procedure. This is an adaptation and extension of a method presented by Lewis and Pramer.[43] Samples were inoculated into a nutritionally balanced, chemically defined, liquid medium containing a relatively high concentration of ammonium sulfate for determination of the ammonia oxidizers, and sodium nitrite for determination of the nitrite oxidizers. After 4 weeks of incubation at room temperature with daily shaking, the number of positive tubes in the ammonia-oxidizing series was determined by analysis of the incubation tubes for nitrite or nitrate nitrogen, or both. The nitrite nitrogen-oxidizing population was determined through analysis of the incubation tubes for nitrate nitrogen after 6 weeks of incubation with daily shaking. The MPN per 100 milliliters was then determined through a standard MPN chart based upon the number of positives in a series of dilutions.

Results. The first three variables studied were the influence on nitrification of initial seed concentration, initial ammonia nitrogen concentration, and the degree of turbulence. During each study, the variable under consideration was investigated over a selected range. Selected results from these three studies are shown in Figures 15 to 17. In these Figures the classical nitrification pattern, oxidation of ammonia nitrogen, and the production of nitrite and nitrate nitrogen is clearly shown. The organic nitrogen concentration was fairly steady throughout the tests. In addition, the excellent recovery of nitrogen throughout the run is demonstrated by the ΣN curve, which is an arithmetic summation of the four forms of nitrogen on each of the days of analyses. As this curve is nearly horizontal, all the nitrogen was continually accounted for throughout the experiment.

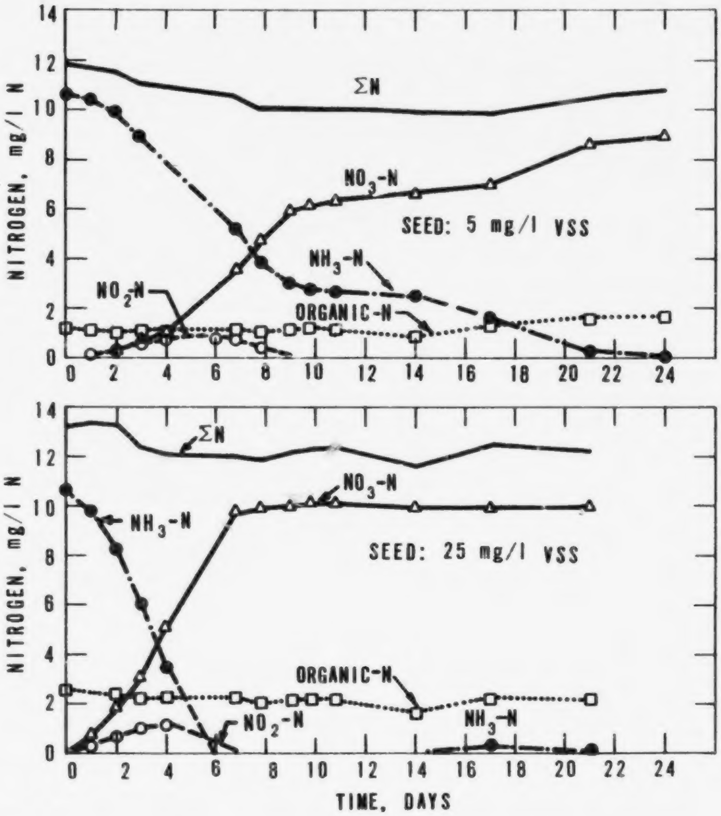

Figure 15. Influence of seed variation on nitrification, NH_3-N constant.

To judge more easily the influence of the variables under study on the rate of oxidation of ammonia nitrogen, Table 1 summarizes the pertinent data from Figures 15 to 17. Table 1 shows that the initial seed concentration influences the rate of ammonia oxidation, as does the degree of turbulence, but that the initial ammonia nitrogen concentration has little influence on this rate. The reason why a fivefold increase in initial seed concentration only causes a twofold increase in the rate of ammonia nitrogen oxidation (Codes A and B) and the reasons why the rate of ammonia nitrogen oxidation in study Code E is much higher than in study Code D is not known at this time (1964).

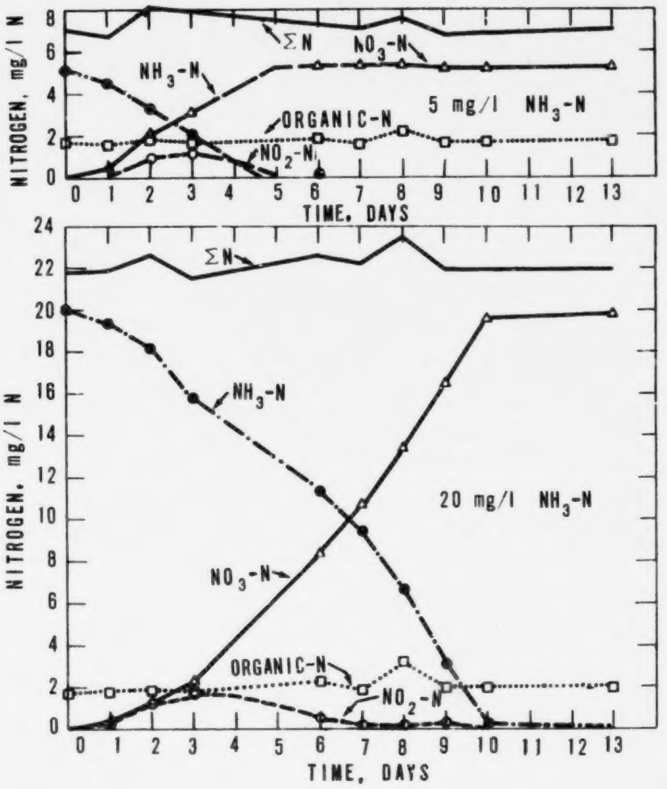

Figure 16. Influence of ammonia variation on nitrification, seed constant.

Table 1. INFLUENCE OF INITIAL SEED CONCENTRATION, INITIAL AMMONIA NITROGEN CONCENTRATION, AND TURBULENCE ON RATE OF AMMONIA NITROGEN OXIDATION

Code	Study	Nominal volatile seed concentration, mg/l	Nominal ammonia nitrogen concentration, mg/l	Turbulence	Rate of ammonia nitrogen oxidation, mg/l per day
A	Seed	5	10	High	1.0
B	Variation	25	10	High	2.2
C	NH_3-N conc.	10	5	High	1.2
D	Variation	10	20	High	1.8
E	Turbulence	10	20	High	7.4
F	Variation	10	20	Low	2.8

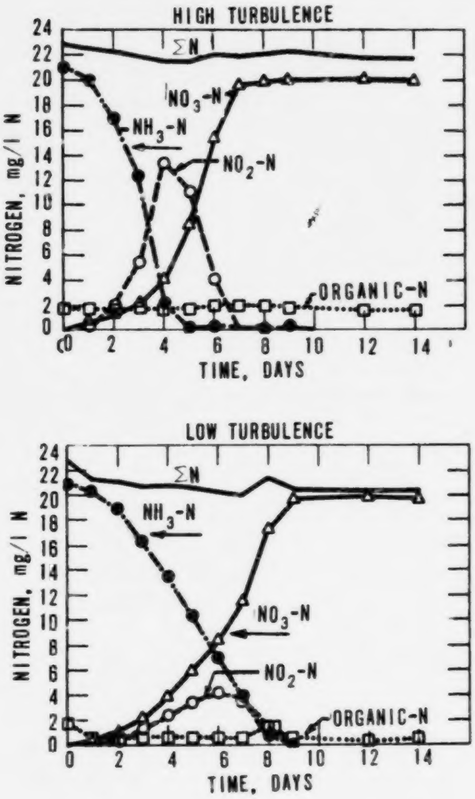

Figure 17. **Influence of turbulence on nitrification.**

The turbulence study indicates that if all other environmental conditions were the same, a flowing stream would show a more rapid oxidation of ammonia nitrogen than would a quiescent body of water such as an impoundment. This is reasonable, as turbulence would tend to keep all the microorganisms in suspension and in intimate contact with the ammonia nitrogen.

Figure 18 shows an example of the correlation of the autotrophic microorganism population with the chemical transformations of nitrogen in the same system. The ammonia-oxidizing population rises rapidly and maintains a comparatively high level during the active oxidation of

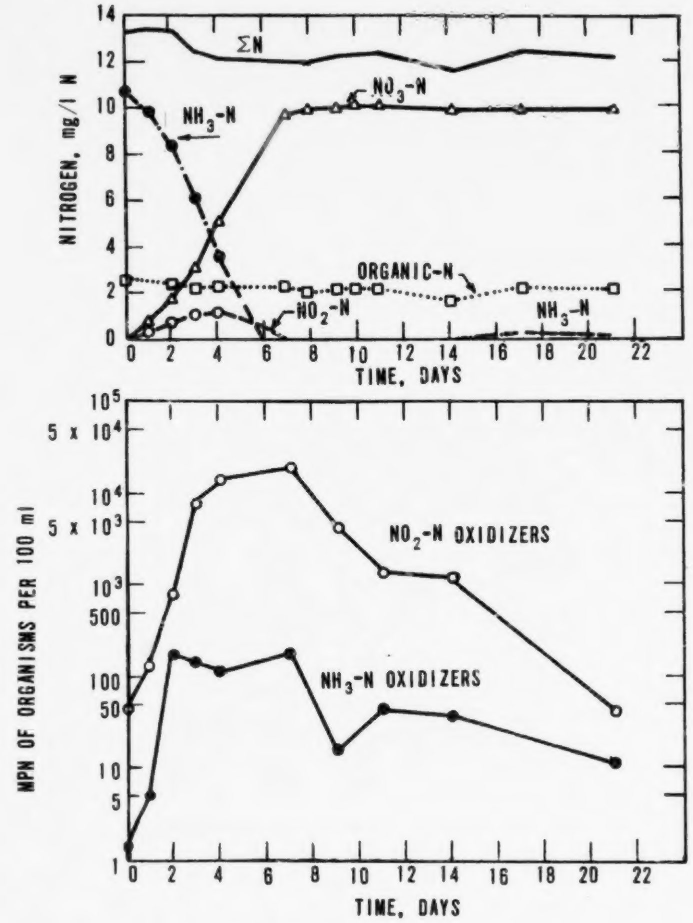

Figure 18. Correlation of bacterial and chemical data during nitrification.

ammonia, but after the sixth day, when the ammonia concentration in the system has reached zero, the ammonia-oxidizing population starts to decrease. The nitrite-oxidizing population rises steadily and peaks on the seventh day, which corresponds to the completion of nitrification. After this time, the nitrite-oxidizing population begins to decline. Excellent correlation was made between the nitrifying populations and the chemical changes that occur during nitrification.

Table 2 shows another interesting correlation that can be made between the biologic populations and the chemical changes that occurred during nitrification. In some of the nitrogen transformation studies, a relatively high concentration of nitrite nitrogen accumulated in the system before it was converted to nitrate nitrogen. A typical example of this is shown by the upper graph in Figure 17. In other studies, as soon as nitrite nitrogen was produced from ammonia nitrogen, it was immediately transformed into nitrate nitrogen, so that no high concentration of nitrite nitrogen ever accumulated. Figures 15 and 16 are examples of this situation. The relative population of ammonia and nitrite oxidizers that existed in the system at the time that nitrification began should determine whether or not nitrite nitrogen accumulated in the system.

Table 2. RELATIONSHIP OF NITRIFYING POPULATION TO ACCUMULATION OF NITRITE NITROGEN DURING NITRIFICATION

Nominal volatile seed concentration, mg/l	Nominal ammonia nitrogen concentration, mg/l	Ammonia nitrogen oxidizing population, MPN per 100 ml	Nitrite nitrogen oxidizing population, MPN per 100 ml	Ratio of nitrite nitrogen population to ammonia nitrogen population	Concentration nitrite nitrogen >60% of original ammonia nitrogen concentration
5	10	49 (3)[a]	23 (5)[b]	0.47	yes
25	10	49 (0)	460 (3)	9.3	no
10	5	17 (0)	27 (3)	1.6	yes
10	25	17 (0)	22 (3)	1.3	yes
10	5	172 (4)	278 (7)	1.6	yes
10	25	700 (3)	542 (4)	0.76	yes
5	10	33 (2)	348 (3)	10.9	no
25	10	<2 (0)	130 (1)	>65	no

[a] The numbers in parentheses indicate the time in days after the start of the test when ammonia nitrogen oxidation began and the ammonia nitrogen oxidizing population was determined.

[b] The numbers in parentheses indicate the time in days after the start of the test when the nitrite oxidizing population was determined. The closest available subsequent count was used; the delays, when they occurred, were because of sampling gaps.

Table 2 shows this to be true. Column 3 shows the ammonia nitrogen-oxidizing population at the time the ammonia oxidation began. Column 4 shows the nitrite nitrogen-oxidizing population on the next available day of analysis. Column 5 shows the ratio of the nitrite nitrogen-oxidizing population to the ammonia nitrogen-oxidizing population. Whenever the ratio was high-9.3 or greater-there was no significant accumulation of nitrite nitrogen in the system, but when the ratio was low-1.6 or less-then nitrite nitrogen did accumulate.

Another interesting finding from these studies was the correlation of the oxidation of ammonia nitrogen with the destruction of alkalinity caused by the production of acidic materials. A typical example is shown in Figure 19, where the pattern of the reduction of alkalinity follows the pattern of the oxidation of ammonia nitrogen exactly. The average ratio of 6.5 milligrams of alkalinity as $CaCO_3$ destroyed per liter per milligram of ammonia nitrogen oxidized per liter held for all the experiments performed thus far. The theoretic destruction of alkalinity is 7.1 milligrams alkalinity destroyed per liter per milligram of ammonia nitrogen oxidized per liter. Alkalinity destruction can, therefore, be used to follow the progress of

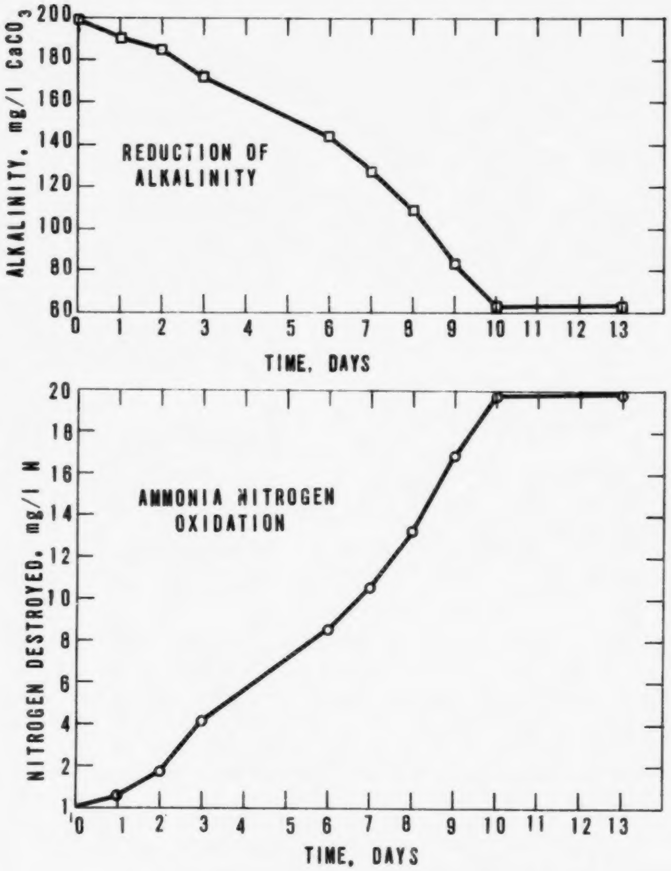

Figure 19. Relation of ammonia nitrogen oxidized to alkalinity destruction.

nitrification, although the nature of the end product, nitrite or nitrate nitrogen, cannot be determined from this analysis alone.

NEED FOR COOPERATION AND COUNSEL

Public Law 660, as amended, instructs the Department of Health, Education, and Welfare to research the problems of the behavior of impoundments and the influence of impoundment releases on downstream water quality, so that intelligent advice may be given to other agencies and valid evaluations made. The law does not insist that the Secretary recommend inclusion of storage capacity in all reservoirs, but does state that the Secretary must be ready to evaluate any streamflow regulation project and his advice must be sought.

The position of the Corps of Engineers was summarized by Weber,[44] who stated:

''... we are trying to work out appropriate and effective arrangements for coordination and distribution of effort among all agencies who can and should contribute to solution of water problems. Insofar as water quality considerations are concerned, we are looking to the Department of Health, Education, and Welfare for both day-to-day advice and continuing, concurrent consideration of long-range studies.''

Burnett[45] also indicated that the Bureau of Reclamation was prepared to seek assistance. He stated:

''In our work in the field of water quality, our experiences working with the Fish and Wildlife Service, the Public Health Service, and the National Park Service have been of paramount benefit. These agencies have the contacts and sources of information not readily available to the Bureau. They also have biological knowledge outside the normal experience of our Reclamation personnel. The use of these sources by our planners has allowed us to progress in water quality control work at an accelerated pace.''

With regard to current water quality control projects, Weber[44] noted:

''The Corps, in 1962, had in operation 214 reservoir projects containing 169 million acre-feet of storage capacity, primarily for flood control and power purposes, with provisions for recreational use at most and with storage for water supply at some. Needs for flow regulation for water quality control were considered in planning many of these projects, but because of the absence of a positive policy for Federal participation in providing for such regulation, specific measures for quality control were included in relatively few projects, such as Fall River, Toronto, and Kanopolis Reservoirs in Kansas; Berlin and Mosquito Creek Reservoirs in Ohio; Youghiogheny River Reservoir in Pennsylvania; and Tygart Reservoir in West Virginia.''

Burnett[45] stated for the Bureau of Reclamation that: ''There are numerous specific project authorizations for construction and operation of Reclamation projects, some of which include works specifically for water quality control.''

The statements above should indicate that, although the research needed in this area is not complete, projects designed to control water quality through streamflow regulation are a reality. Past research on the behavior of impounded water is very helpful to these projects. This, together with accumulating experience in the operation of these projects, has allowed a better understanding of impoundments, which has improved the estimates of effects on water quality and the designs of projects. But, as this article has tried to indicate, to understand completely the behavior of impoundments and their influences on downstream water quality, additional research work still needs to be done. In some instances the needed research involves integrating existing information and filling in gaps. In some instances it involves measuring effects in impoundments and streams that have been measured in other circumstances. In some instances, further research must relate the environment to the measured effect. In other instances, whole new areas of interest must be researched.

SUMMARY

To begin to fill the research needs of the phase of the overall project concerned with the behavior of water quality in impoundments, the staff of the engineering section at the Robert A. Taft Sanitary Engineering Center started studies in five major areas:

1. The simulation of stratified impoundments on a laboratory scale.

2. The rate and extent of the dissolution of iron and manganese from bottom muds in stratified impoundments.

3. The DO budget of impoundments.

4. The possibility of artificial destratification of impoundments.

5. The biologic transformations of nitrogen in impoundments.

As these studies have only recently begun, no definitive conclusions can be made at this time. Preliminary results are, however, encouraging and indicate that these projects are worthwhile and potentially productive, and will eventually lead to the increased understanding of the behavior of impoundments.

In addition to the studies on the behavior of water quality in impoundments, attention is being given to alternative methods of solving water pollution problems. A project designed to provide construction and operating costs for water treatment is being explored. If these figures were accurately known, the additional cost of treatment for the removal of a given pollutant could be compared with the cost of providing dilution water. Moreover, the possibility of aerating a stream mechanically, instead of supplying oxygen to a critical zone with impoundment releases, is being considered. One million gallons of water with a DO concentration of 8 milligrams per liter contains only 66.5 pounds of oxygen, so stream aeration may be a less expensive method of providing this oxygen. These alternatives are important and must be considered, as construction of impoundments and streamflow regulation alone will not solve basin-wide pollution problems. Most impoundments actually reduce the annual flow because of evaporation and other associated water losses, although they can be of great benefit when properly used.

Although the research goals of the overall projects are broad, in general, they are designed to measure the influences of impounding water on water quality, to relate these influences to the environment to facilitate prediction of the influences of new impoundments on water quality and the influences of releases on the downstream water quality, or to develop operational tests that will permit these predictions. The predictions of the behavior and influence of new impoundments will allow alternative sites, alternative methods of operation, or alternative treatment plans to be accurately integrated into an economic basin-wide analysis. Only through optimization of basin designs, using improved mathematical and economic techniques with better water quality behavior input data and realistic alternative proposals, can streamflow regulation projects attain the maximum degree of water quality control.

ACKNOWLEDGMENTS

The authors wish to thank the other principal staff members of the engineering section of the Robert A. Taft Sanitary Engineering Center who were assigned to the project ''The Effects of Streamflow Regulation on Water Quality.'' They are: William H. Irwin, Jack DeMarco, Jerzy Kurbiel, Andre Cobet, and Jesse M. Cohen. Each of these staff members contributed to the conception of the investigations described in this article, the taking of the data presented, and the preparation of the final manuscript.

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PART II

INVESTIGATIVE
RESEARCH
PAPERS

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PAPER NO. 3

Discussion of

"EFFECT OF BENTHIC ALGAE ON STREAM DISSOLVED OXYGEN"*

J.M. Symons

This discussion of a paper by Richard L. O'Connell and Nelson A. Thomas, covers the first phase of the study on the dissolved oxygen budget in reservoirs. The development of the apparatus, the recording circuitry, and some early results are contained herein. A report on the remainder of this research is in the paper that follows.

The correct reference citation is:

Symons, J.M. "Discussion of 'Effect of Benthic Algae on Stream Dissolved Oxygen' by O'Connell and Thomas." Journal Sanitary Engineering Division, Proceedings of the American Society of Civil Engineers, 92, SA-1, 301-306. February 1966.

O'Connell and Thomas are to be congratulated on an interesting and useful piece of work. The use of a chamber for the direct evaluation of the influence of benthic algae on the dissolved oxygen (DO) budget of a stream is a valuable contribution. The direct measurement approach is particularly interesting to the writer, who, as project leader of the United States Public Health Service's research on ''Effect of Streamflow Regulation on Water Quality,'' has given considerable attention to the related problem of direct measurement of changes in the DO budget in impoundments. Despite the utility of the apparatus described, the writer would like to suggest that its

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*O'Connell, R. L., and N. A. Thomas. J. Sanit. Eng. Div., Proc. ASCE. 91(SA3): 1-16. June 1965.

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operation might be improved if DO probes were included in the chamber. This would yield a continuous record of the DO concentration changes, rather than depending on periodic withdrawal of a sample for wet chemical analysis. The Engineering Research Section personnel at the Robert A. Taft Sanitary Engineering Center, Cincinnati, Ohio, have, for some time, been developing an apparatus involving probes for evaluating the DO budget in impoundments.[1] In an effort to show the general usefulness of this approach, this apparatus is described in detail below.

The importance of algae in the DO budget of polluted waters has been extensively reported by Hull,[2,3,4] among others. In most of his work, Hull has used the light- and dark-bottle technique for evaluating photosynthetic oxygenation, although he is well aware of the limitations of this technique. The five most commonly noted inaccuracies in the bottle technique are as follows: (1) An unnaturally high surface-to-volume ratio in the bottles tends to increase the number of organisms in the bottles over that which exists in the natural environment; (2) the relatively long time the bottles must be left in place to permit the detection of changes in DO concentrations may cause problems; if the external environment, wind or sunlight, changes significantly during this wait, the results obtained may not be applicable to the general case; (3) there is little or no mixing in the bottles, and this interferes with the flow of food to and the waste products away from the organisms; this may influence their metabolism and make the results somewhat atypical; (4) bottles are usually filled on the boatdeck after water has been withdrawn from the impoundment; during this handling there is the possibility of injuring the algal population by exposing them to sunlight, which can be minimized by filling the bottles in the shade or at night; but, unless an impoundment is well-mixed, there is the danger that when the bottles are returned to the test level for incubation, the water inside the bottles will not truly be representative of the water at the test level; and (5) although the bottle technique does give some measure of the contribution of algal photosynthesis to the oxygen budget, it does little to measure the contribution of atmospheric reaeration to this budget.

The apparatus discussed below, a part of which is pictured in Figure 1, was designed to overcome the inaccuracies inherent in the light- and dark-bottle technique. Briefly, the apparatus consists of three DO probes: One is placed in a dark container; one is placed in a con-

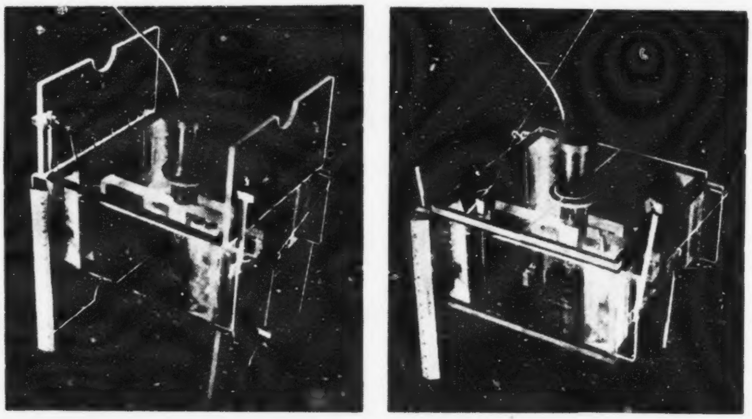

Figure 1. ''Light'' container and DO probe in open and closed position - note 12-inch scale in left foreground.

tainer that admits sunlight; and one remains unconfined in the impoundment water. The probes, which are manufactured by Pro-Tech, Inc.,* are made of silver and carbon, covered with 0.0005-inch-thick Teflon as the membrane. The voltage applied to the probe tip is 1.1 volts, and the DO concentration is indicated on a microammeter. The microammeter dial is adjusted so that it reads DO concentration directly, and the system is temperature-compensated in the range of 15° to 30°C. For operation below 15°C, direct reading may be obtained by turning another dial to a value taken from a previously established calibration curve. The meter box contains four meters, which continuously indicate the temperature of the unconfined water, as well as the DO concentration in the three environments.

To use one recorder and to permit increased sensitivity of the DO readings, the circuit shown in Figure 2 was developed by the Model Shop at the Robert A. Taft Sanitary Engineering Center. Essentially, the microampere output of the meters was converted to millivolts through the use of potentiometers. The size of these potentiometers would depend on the output of the particular DO probe system used. For this system, a 500-ohm potentiometer and a 100-

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*Similar DO probes currently (1969) manufactured by Automated Environmental Systems, Inc., Woodbury, Long Island, New York.

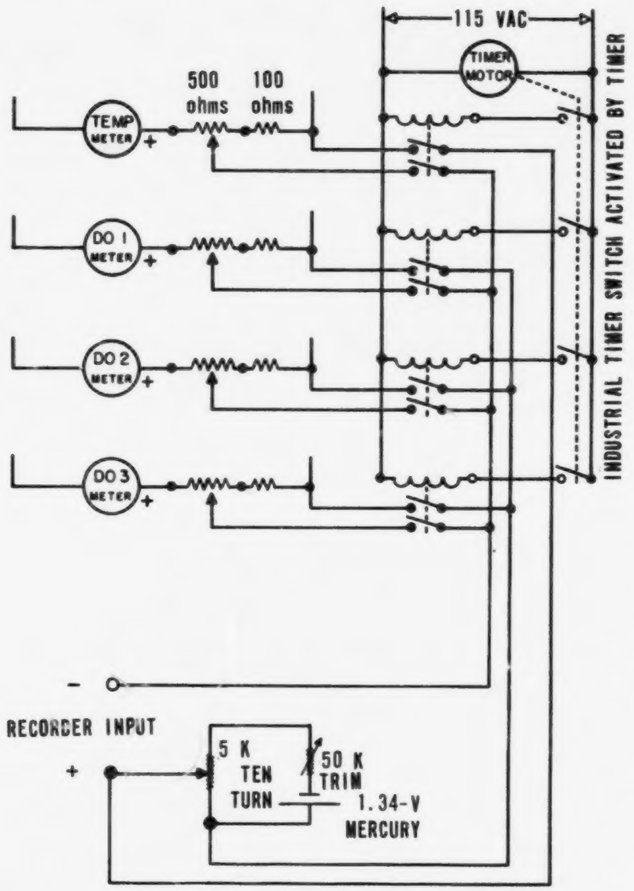
115 VAC
TIMER MOTOR
500 ohms
100 ohms
TEMP METER
+
DO 1 METER
+
DO 2 METER
+
DO 3 METER
+
INDUSTRIAL TIMER SWITCH ACTIVATED BY TIMER
-
RECORDER INPUT
+
5 K TEN TURN
50 K TRIM
1.34-V MERCURY

Figure 2. **Circuit diagram for switching and recording output of temperature and three DO probes, and for bucking the DO probes' output - traverse motor switching circuit not shown.**

ohm fixed resistor were used. A cam-type switch on a 1/2-hour cycle was used to record each of the four meter readings in sequence; therefore, a 7-1/2-minute record of temperature and of each of the three DO probe readings is obtained on the recorder every 1/2 hour.

Because the motors for driving the propellors that circulate water past the DO probes are battery-operated, they too are switched on and off periodically to extend the battery life. Running the motor for 10 minutes prior to recording a reading is sufficient to bring the probe to the proper value. This part of the circuit is not shown in Figure 2.

Finally, a bucking voltage circuit was used so that a 0- to 10-millivolt recorder could be used, and the system could always be ''on scale'' for any DO concentration reading. The temperature circuit was not bucked, however. Because the full-scale output of the DO probe system is 50 millivolts, and this is equivalent to 15 milligrams per liter of DO, the use of a 10-millivolt recorder means that the full-scale deflection of the recorder, 11 inches in this case, is equivalent to 3 milligrams per liter of DO. With this recording system, a change of 0.1 milligram per liter of DO concentration results in a pen deflection of 0.4 inch, and the use of the bucking voltage circuit allows the output of the probes to be ''on scale'' regardless of the original DO concentration.

The sensitivity and reliability of this apparatus can be seen from Figure 3. In this experiment all three probes were suspended in a tank of water with the doors of both containers open throughout the test. Duplicate Winkler analyses showed that the DO concentration dropped from 8.4 to 7.6 milligrams per liter throughout the 17-hour test. Figure 3 is a plot of the recorder readings for each of the three DO probes. These data show that, despite an average change in DO concentration of only 0.045 milligram per liter per hour, only 3 or 4 hours of data collection was necessary to establish this rate. For comparison, the magnitudes of the recorder deflection for 1 milligram per liter and 0.1 milligram per liter of DO concentration are shown in Figure 3.

The heart of the apparatus, the three probes in their different environments, was designed specifically to avoid some of the inaccuracies inherent in the light- and dark-bottle technique. First, the surface-to-volume ratio in the plastic boxes is nearly one-third that of a standard 300-milliliter bottle. Second, continuous measurement of the changes in DO concentration will allow the experiment to be conducted over the minimum time necessary to detect these changes. This will permit testing under as nearly constant conditions of weather as possible. Third, because water must flow past a DO probe for it to function properly, the contents of the plastic containers will be mixed by the action of the propellor creating the flow past the

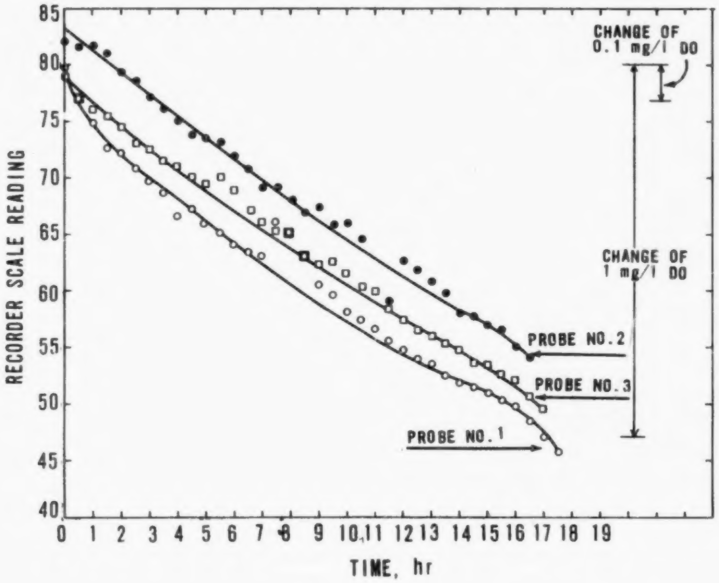

Figure 3. DO concentration data taken with all probes in the same environment.

probes. This will help maintain an adequate flow of food to the organisms in the containers. Fourth, by lowering the containers with the trapdoors open and then closing them by tripping spring-loaded triggers, a relatively undisturbed sample of water will be trapped inside the containers from the depth at which the experiment is occurring. This should avoid any possible upset that might be caused by transporting the organisms to the surface. Fifth, because one probe will be in water not confined in any container, the influence of atmospheric reaeration on the DO budget in the impoundment can also be investigated. Of course, some of the problems inherent in ''bottle'' studies are not eliminated through the use of this equipment, but it is a significant improvement.

In summary, the data that can be obtained with this apparatus can be related to a form of Equation 5 in the original paper

$$\frac{dc}{dt} = (P-R) - k_1L_a + k_2D$$

in which dc/dt = rate of change of DO concentration in unconfined water with time; P = rate of gross oxygen production by algae; R = rate of oxygen uptake from algal respiration; P-R = rate of net oxygen production by algae; k_1L_a = rate of oxygen uptake from bacterial metabolism of organic matter; k_2D = rate of oxygen brought into solution from the atmosphere, and D is the DO deficit. The DO probe in unconfined water gives a measure of the term ''dc/dt.'' The probe inside the container that admits sunlight gives a measure of ''$(P-R) - k_1L_a$,'' whereas the DO probe in the dark container evaluates ''$R + k_1L_a$.'' The terms P and k_2D can then be obtained by appropriate differences.

The writer again wishes to compliment the authors on the development of a useful device for direct evaluation of algal contribution to the DO budget. This discussion has been prepared and the apparatus described with the hope that it might stimulate others to attempt the direct evaluation of the factors affecting the DO budget, using this type of apparatus or a similar one.

ACKNOWLEDGMENTS

The writer wishes to acknowledge the efforts of the personnel of the Model Shop at the Robert A. Taft Sanitary Engineering Center, Cincinnati, Ohio, Joseph A. Castelli, Supervisor, and Reuben Daily, A. B. Arnold, and Frank J. Reis, who helped to develop the equipment described herein.

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PAPER NO. 4

MANAGEMENT AND MEASUREMENT OF DO IN IMPOUNDMENTS

(Investigative Portion)

J.M. Symons, W.H. Irwin, R.M. Clark,
and G.G. Robeck

Paper No. 4 as originally published combined two studies, one classed herein as investigative, the other as engineering. The investigation, reported here, followed completion of the research discussed in the previous paper, "Discussion of Effect of Benthic Algae on Stream Dissolved Oxygen," and concerns the measurement of the dissolved oxygen budget in reservoirs. Paper No. 4 - Part 2, numbered Paper 9 in this Compilation, deals with the engineering phase of the study and appears in the Engineering Research section of this document. Data were taken with the previously described equipment, and a mathematical model was proposed. A solution to the proposed model is also presented. The Figures, Tables, and References have been renumbered for convenience. A paper similar to this, but with a less accurate solution to the mathematical model appeared in the Proceedings of the National Symposium on Quality Standards for Natural Waters held at the University of Michigan in 1966. Three discussions and a closure are also reprinted.

The correct reference citations are:

Symons, J. M., W. H. Irwin, R. M. Clark, and G. G. Robeck. "Management and Measurement of DO in Impoundments." Journal Sanitary Engineering Division, American Society of Civil Engineers, 93, SA6, 181-209. December 1967.

Thackston, E. L. Ibid. 94, SA4, 748-52. August 1968.
Bella, D. A. Ibid. 94, SA5, 1034-37. October 1968.
Hull, C. H. J. Ibid. 95, SA1, 158-78. February 1969.
Closure. Ibid. 95, SA2, 342-49. April 1969.

THEORETICAL CONSIDERATIONS

Mathematical Model

One common mathematical model used to describe the change in dissolved oxygen (DO) concentrations with time in the euphotic zone of impoundments, where sedimentation is not an important factor, is shown in Equation (1), (see O'Connell and Thomas[1]),

$$\frac{dD}{dt} = k_1 \, (L_\alpha - y) - k_2 \, (D) + (R - P) \tag{1}$$

in which

D = DO deficit at any time,
t = time,
k_1 = deoxygenation rate constant,
L_α = total organic biochemical oxygen demand,
y = biochemical oxygen demand satisfied at any time $\{y = L_\alpha \, [1 - \exp(-k_1 \, t)]\}$,
k_2 = atmospheric reaeration rate constant,
R = rate of oxygen demand by the algal population,
and P = rate of oxygen produced by the algal population.

Integration yields

$$D = \frac{k_1 \, (L_\alpha)}{k_2 - k_1} \, [\exp(-k_1 \, t) - \exp(-k_2 \, t)] + \frac{(R - P)}{k_2} \, [1 - \exp(-k_2 \, t)] + D_\alpha \, [\exp(-k_2 \, t)] \tag{2}$$

in which D_α = initial DO deficit.

Although Hull's[2] suggestion of measuring all the DO sources and sinks in a given impoundment to evaluate the DO budget is valid, this is often time-consuming and the budget is evaluated only under one set of environmental conditions. If a reliable model can be developed, the changes in the constants, as related to changing sunlight, wind, algal counts, and nutrient level, will permit general conclusions to be drawn, permit predictions to be made, and allow comparisons of one impoundment with another.

Through the years, attempts have been made to devise methods to evaluate and understand the physical meaning of the five constants (k_1, k_2, L_a, R, and P) in Equation (2). While excessive concern over the precise meanings of these constants has, with some justification, been criticized by Hull[2] and Ettinger,[3] this does not negate the utility of the overall model. The important feature of the model as used here is not the physical interpretation of the constants, but its ability to reproduce accurately what is occurring in nature. For this analysis Equation (2) might be rewritten,

$$D = \frac{ac}{b-a}\left[\exp(-at) - \exp(-bt)\right] + \frac{(d-f)}{b}\left[1-\exp(-bt)\right] + D_a\left[\exp(-bt)\right] \qquad (3)$$

in which a, b, c, d, and f correspond mathematically to k_1, k_2, L_a, R, and P, respectively.

The model shown in Equation (2) may not be an exact representation of the complete DO budget in an impoundment, but, as shown later in Figure 2, the change in D with time derived from the solution of the model, and the change in the dissolved oxygen deficit with time in the natural environment, are similar. Therefore, the model is acceptable for further use.

Evaluation of the Model by the DO Probe Method

There are many different methods of studying the DO budget in natural waters; for example, those of Odum[4] (diurnal cycle method); Hull[5,6] (light and dark bottle method); Goldman and Carter[7] (carbon-14 method); and Verduin[8] (changes in CO_2 concentration in unconfined water method). Each method has advantages and drawbacks. The method developed herein, called the DO probe method, using special equipment has three distinct advantages: (1) The inaccuracies encountered in bottle experiments are minimized; (2) DO is measured directly; and (3) the model, Equation (2), may be solved from data gathered.

Future study will undoubtedly reveal further improvements in this approach, but this is a significant advance.

Details of the design and operation of this equipment, and a discussion of its advantages over the bottle technique, have been given by Symons[9] and are noted only briefly here. The apparatus consists of three DO probes that will record the changes in DO concentration with time under three different conditions. One probe is placed in a black container that prevents the admission of either sunlight or air. Under these conditions the constants k_2 and P are zero in Equation (1). Integrating Equation (1) under these conditions yields

$$D = L_a\,[1-\exp(-k_1 t)] + R\,(t) + D_a \tag{4}$$

A second DO probe is in a clear plastic container that admits sunlight, but not air. Under these conditions, Equation (1) can be integrated with $k_2 = 0$. This integration yields

$$D = L_a\,[1-\exp(-k_1 t)] + (R-P)\,(t) + D_a \tag{5}$$

The third DO probe is unconfined; therefore, the change in DO deficit with time is represented by Equation (1), which when integrated yields Equation (2).

Evaluation of the five constants in the model is accomplished by solving Equation (4) for k_1, R, and L_a using DO versus time data obtained from the DO probe in the dark container. These constants are then entered in Equation (5) and P is evaluated through the use of the DO versus time data taken from the DO probe in the light container. Finally, Equation (2) is solved from the DO data taken with the probe that is unconfined, and the previously developed constants.

For those interested in the details of the solution, an example is given in the Appendix. The application of this method, using data from the equipment developed for this purpose[9] will be demonstrated in the results section herein.

EQUIPMENT AND METHODS

The ability of the special DO probe equipment to produce data necessary for the solution of Equations (2), (4) and (5) was demonstrated by studies with a 750-gallon tank filled with artificial impoundment water. The simulated impoundment water was created by mixing some settled sewage, nitrogen, phosphorus, sodium bicarbonate, and algal seed with Cincinnati tap water. This mixture was stirred 15 minutes each hour and illuminated 14 hours each day until an observable algal growth developed. On the day of the test, some sewage was added as a fresh source of organic material. The three DO probes, with the trap doors on the containers open, were lowered into the tank, the triggers were tripped, and the test begun. The DO concentration data were obtained for an 8-hour period, under illumination, using the recording and timing circuitry detailed by Symons.[9] Before the test began, the DO probes were calibrated and adjusted against the Winkler method for DO analysis. The temperature of the test was fairly constant at 20°C.

RESULTS

The data plotted in Figure 1 were obtained with the special equipment previously described.[9] Some important

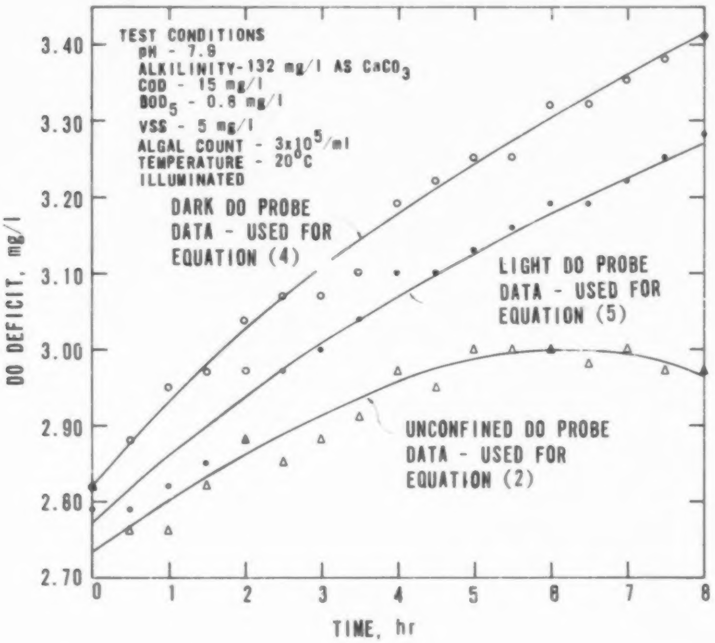

Figure 1. Data from special DO probe equipment.

water quality parameters of the artificial impoundment water used for the test are noted on Figure 1, which also shows the excellent data that may be obtained using this equipment (note the extremely expanded ordinate scale). Approximate curves showing the trends have been placed by eye through the data points.

The solutions to Equations (2), (4), and (5) are derived from the deficit data taken from these curves. Table 1 summarizes the data used for the solution of the equations. Using these data and the method in the Appendix the solution for Equation (2) is

$$D = \frac{(0.31)\,(0.51)}{0.01 - 0.31}\{\exp[(-0.31)\,(t)] - \exp[(-0.01)\,(t)]\}$$

$$+ \frac{-0.01}{0.01}\{1 - \exp[(-0.01)\,(t)]\} + 2.74\{\exp[(-0.01)\,(t)]\} \qquad (6)$$

Table 1. DATA FROM FIGURE 1 FOR THE SOLUTION OF THE MATHEMATICAL MODEL

	DO deficit, milligrams per liter		
Time, hr (1)	Dark probe (2)	Light probe (3)	Unconfined probe (4)
0	2.82	2.77	2.74
0.5	2.88	2.82	2.78
1.0	2.94	2.86	2.81
1.5	2.99	2.90	2.84
2.0	3.03	2.94	2.87
2.5	3.07	2.98	2.90
3.0	3.11	3.01	2.92
3.5	3.14	3.04	2.95
4.0	3.18	3.08	2.97
4.5	3.21	3.10	2.99
5.0	3.24	3.13	3.00
5.5	3.27	3.16	3.01
6.0	3.30	3.19	3.01
6.5	3.33	3.21	3.01
7.0	3.36	3.23	3.00
7.5	3.39	3.25	2.99
8.0	3.41	3.27	2.98

Figure 2 compares the line of best fit from Equation (6) with the deficit data obtained with the unconfined DO probe. The close correlations of the data generated with the mathematical model and the data taken with the DO probe shows the general reliability of the model. The low values of k_2 = 0.01, and P = 0.06, are in agreement with the low turbulence and low algal population in the test

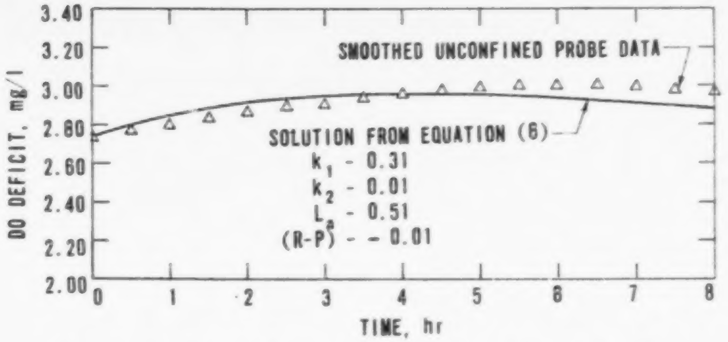

Figure 2. Comparison of Equation (6) with the smoothed unconfined probe data.

media. These data show that the DO probe method, using this special equipment and the mathematical model represented by Equation 2, is a useful method for studying the DO budget in impoundments.

FUTURE PLANS

The DO probe equipment must be used in the field to experience the influence of various natural environmental factors on the constants in the mathematical model, and to determine whether good correlations between the model and data taken may be obtained under a variety of conditions.

SUMMARY AND CONCLUSIONS

To evaluate the DO budget in impoundments properly, an adequate mathematical model and a method of solving the model are needed. The model presented in Equation (2) is adequate, and the DO data obtained with the special three-probe system[9] are effective for determining the five constants. Use of this tool will permit further study of the DO budget and will permit comparisons of the DO budgets of various impoundments. These specific conclusions may be drawn:

1. Equation (2) is a satisfactory mathematical model for representing the DO budget in impounded waters.
2. The special three-DO-probe equipment designed for this study is capable of supplying the data necessary for solving the DO model.

APPENDIX - MATHEMATICAL MODEL*

The assumed constants for this example are:

k_1 = 0.10 per hour,
k_2 = 0.05 per hour,
R = 0.10 per hour,
P = 0.30 per hour,
L_α = 3.00 milligrams per liter,
D_α = 0.47 milligram per liter.

Using the appropriate constants and solving Equations (2), (4), and (5) yields the deficit data shown in Table 2, and plotted in Figure 3. For simplicity, the units of the constants are not used.

Several approaches to the solution of this problem were attempted. Various forms of exponential approximation or Prony's method[10] and simple iteration were tried without success. Much of the difficulty in using these techniques was caused by an insufficient number of significant figures and error caused by rounding off. The most productive technique was a form of the Newton Raphson Method[11] which will be developed using the data in Table 2. A computer program for the solution presented was written in Fortran II for use with a Honeywell 400 computer.

The first objective was to find a best fit to Equation (7), which is Equation (4) in slightly different form,

$$y_i = D_i - D_a = L_a\,[1-\exp(-k_1\,t_i)] + R\,t_i \tag{7}$$

Expanding y_i in a Taylor Series about the point L_α^0, k_1^0, R^0 yields

$$y_i = L_a^0\,[1-\exp(-k_1^0\,t_i)] + R^0\,t_i + \frac{\partial y_i}{\partial L_a}\,\Delta\,L_a + \frac{\partial y_i}{\partial k_1}\,\Delta\,k_1 + \frac{\partial y_i}{\partial R}\,\Delta\,R \tag{8}$$

neglecting second-order terms. The superscripts on the variables L_α, k_1, and R indicate the initial approximation, and $\Delta L_\alpha = L_\alpha - L_\alpha^0$, $\Delta k_1 = k_1 - k_1^0$, and $\Delta R = R - R^0$. Evaluating the partial derivatives yields

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*As originally published this Appendix contained a serious error. This error was discovered by Dr. C. H. J. Hull of the Delaware River Basin Commission, who published a correction in his Discussion of this paper. The Appendix printed here has been corrected, and the authors are indebted to Dr. Hull for his correction.

Table 2. DO DEFICIT DATA FROM ASSUMED CONSTANTS

	DO deficit, milligrams per liter[a]					
	Equation (4) Simulated dark probe[b] (2)		Equation (5) Simulated light probe[c] (3)		Equation (2) Simulated unconfined probe[d] (4)	
Time, hr (1)	2-place data	5-place data	2-place data	5-place data	2-place data	5-place data
0	0.47	0.47000	0.47	0.47000	0.47	0.47000
0.5	0.67	0.66631	0.52	0.51631	0.50	0.50412
1.0	0.86	0.85548	0.56	0.55548	0.53	0.53034
1.5	1.04	1.03787	0.59	0.58787	0.55	0.54918
2.0	1.21	1.21387	0.61	0.61381	0.56	0.56129
2.5	1.38	1.38360	0.63	0.63360	0.57	0.56898
3.0	1.55	1.54754	0.65	0.64754	0.57	0.56671
3.5	1.71	1.70593	0.66	0.65593	0.56	0.56101
4.0	1.86	1.85904	0.66	0.65904	0.55	0.55018
4.5	2.01	2.00711	0.66	0.65711	0.53	0.53472
5.0	2.15	2.15041	0.65	0.65041	0.51	0.51486
5.5	2.29	2.28915	0.64	0.63915	0.49	0.49100
6.0	2.42	2.42357	0.62	0.62357	0.46	0.46352
6.5	2.55	2.55385	0.60	0.60385	0.43	0.43259
7.0	2.68	2.68023	0.58	0.58023	0.40	0.39856
7.5	2.80	2.80289	0.55	0.55289	0.36	0.36175
8.0	2.92	2.92201	0.52	0.52201	0.32	0.32227

[a] D_a = 0.47 milligram per liter.
[b] $k_1 = 0.10$, $k_2 = 0$, $P = 0$, $R = 0.10$, $\overline{L_a} = 3.00$.
[c] $k_1 = 0.10$, $k_2 = 0$, $P = 0.30$, $R = 0.10$, $\overline{L_a} = 3.00$.
[d] $k_1 = 0.10$, $k_2 = 0.05$, $P = 0.30$, $R = 0.10$, $\overline{L_a} = 3.00$.

$$\frac{\partial y_i}{\partial L_a} = [1-\exp(-k_1^0\, t_i)] \qquad (9)$$

$$\frac{\partial y_i}{\partial k_1} = t_i\, L_a^0\, [\exp(-k_1^0\, t_i)] \qquad (10)$$

and

$$\frac{\partial y_i}{\partial R} = t_i \qquad (11)$$

Substituting Equations (9), (10), and (11) into Equation (8) yields

$$y_i = L_a^0\,[1-\exp(-k_1^0\, t_i)] + R^0\, t_i + [1-\exp(-k_1^0\, t_i)]\,\Delta L_a + t_i\, L_a^0\,[\exp(-k_1^0\, t_i)]\,\Delta k_1 + t_i\,\Delta R \qquad (12)$$

Forming the residual equations yields

$$V_i = L_a^0\,[1-\exp(-k_1^0\, t_i)] + R^0\, t_i + [1-\exp(-k_1^0\, t_i)]\,\Delta L_a + t_i\, L_a^0\,[\exp(-k_1^0\, t_i)]\,\Delta k_1 + t_i\,\Delta R - y_i \qquad (13)$$

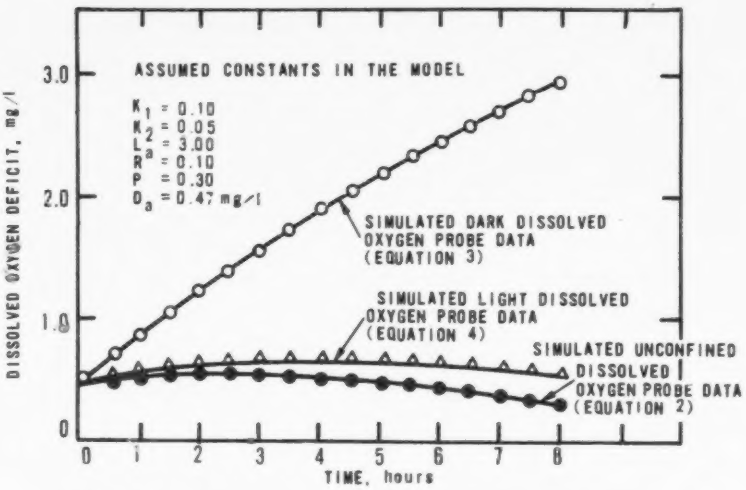

Figure 3. **Assumed dissolved oxygen data to verify solution of dissolved oxygen model,**

Assuming least squares yields

$$S = \sum_{i=1}^{n} V_i^2 = \sum_{i=1}^{n} \{ L_a^0 [1-\exp(-k_1^0 t_i)] + R^0 t_i + [1-\exp(-k_1^0 t_i)] \Delta L_a + t_i L_a^0 [\exp(-k_1^0 t_i)] \Delta k_1 + t_i \Delta R - y_i \}^2 \tag{14}$$

Minimizing S with respect to the parameters ΔL_α, Δk_1, and ΔR, produces

$$\frac{\partial S}{\partial \Delta L_a} = 2 \sum_{i=1}^{n} \{ L_a^0 [1-\exp(-k_1^0 t_i)] + R^0 t_i + [1-\exp(-k_1^0 t_i)] \Delta L_a + t_i L_a^0 [\exp(-k_1^0 t_i)] \Delta k_1 + t_i \Delta R - y_i \} \{[1-\exp(-k_1^0 t_i)]\} = 0 \tag{15}$$

$$\frac{\partial S}{\partial \Delta k_1} = 2 \sum_{i=1}^{n} \{ L_a^0 [1-\exp(-k_1^0 t_i)] + R^0 t_i + [1-\exp(-k_1^0 t_i)] \Delta L_a + t_i L_a^0 [\exp(-k_1^0 t_i)] \Delta k_1 + t_i \Delta R - y_i \} \{ t_i L_a^0 [\exp(-k_1^0 t_i)]\} = 0 \tag{16}$$

and $$\frac{\partial S}{\partial \Delta R} = 2 \sum_{i=1}^{n} \{ L_a^0 [1-\exp(-k_1^0 t_i)] + R^0 t_i + [1-\exp(-k_1^0 t_i)] \Delta L_a + t_i L_a^0 [\exp(-k_1^0 t_i)] \Delta k_1 + t_i \Delta R - y_i \} \{ t_i \} = 0 \tag{17}$$

Dividing through Equation (15) by 2 and multiplying all the terms together yields

$$\begin{aligned}&\sum_{i=1}^{n} \{ L_a^0 \, [1\text{-}\exp(-k_1^0 \, t_i)]^2 + R^0 t_i \, [1\text{-}\exp(-k_1^0 \, t_i)] \\ &+ [1\text{-}\exp(-k_1^0 \, t_i)]^2 \, \Delta L_a + t_i \, L_a^0 \, [\exp(-k_1 \; t_i)] \, [1\text{-}\exp(-k_1 \; t_i)] \, \Delta k_1 \\ &+ t_i \, [1\text{-}\exp(-k_1^0 \, t_i)] \, \Delta R - y_i \, [1\text{-}\exp(-k_1^0 \, t_i)]\} = 0\end{aligned} \tag{18}$$

Performing a similar operation on Equation (16) and Equation (17) yields

$$\begin{aligned}&\sum_{i=1}^{n} \{ L_a^0 \, [1\text{-}\exp(-k_1^0 \, t_i)] \, [t_i \, L_a^0 \exp(-k_1^0 \, t_i)] + R^0 \, t_i^2 \, L_a^0 \, [\exp(-k_1^0 \, t_i)] \\ &+ t_i \, L_a^0 \, [\exp(-k_1^0 \, t_i)] \, [1\text{-}\exp(-k_1^0 \, t_i)] \, \Delta L_a \\ &+ t_i^2 \, (L_a^0)^2 \, [\exp(-2 \, k_1^0 \, t_i)] \, \Delta k_1 \\ &+ t_i^2 \, L_a^0 \, [\exp(-k_1^0 \, t_i)] \, \Delta R - y_i \, t_i \, L_a^0 \, [\exp(-k_1^0 \, t_i)]\} = 0\end{aligned} \tag{19}$$

and
$$\begin{aligned}&\sum_{i=1}^{n} [L_a^0 \, [1\text{-}\exp(-k_1^0 \, t_i)] \, t_i + R^0 \, t_i^2 + [1\text{-}\exp(-k_1^0 \, t_i)] \, t_i \, \Delta L_a \\ &+ t_i^2 \, L_a^0 \, [\exp(-k_1^0 \, t_i)] \, \Delta k_1 + t_1^2 \, \Delta R - y_i \; t_i\} = 0\end{aligned} \tag{20}$$

Rearranging Equations (18), (19), and (20) yields

$$\begin{aligned}&\sum_{i=1} [1\text{-}\exp(-k_1^0 \, t_i)]^2 \, \Delta L_a \\ &+ \sum_{i=1}^{n} \{t_i \, L_a^0 \, [\exp(-k_1^0 \, t_i)] - t_i \, L_a^0 \, [\exp(-2 \, k_1^0 \, t_i)]\} \, \Delta k_1 \\ &+ \sum_{i=1}^{n} \{t_i - t_i \, [\exp(-k_1^0 \, t_i)]\} \, \Delta R = - \sum_{i=1}^{n} L_a^0 \, [1\text{-}\exp(-k_1^0 \, t_i)]^2 \\ &- \sum_{i=1}^{n} R^0 \, t_i \, [1\text{-}\exp(-k_1^0 \, t_i)] + \sum_{i=1}^{n} y_i \, [1\text{-}\exp(-k_1^0 \, t_i)]\end{aligned} \tag{21}$$

$$\begin{aligned}&\sum_{i=1}^{n} \{t_i \; L_a^0 \, [\exp(-k_1^0 \, t_i)] - t_i \, L_a^0 \, [\exp(-2 \, k_1^0 \, t_i)] \} \, \Delta L_a \\ &+ \sum_{i=1}^{n} \{t_i^2 \, (L_a^0)^2 \, [\exp(-2 \, k_1^0 \, t_i)]\} \, \Delta k_1 \\ &+ \sum_{i=1}^{n} \{t_i^2 \, L_a^0 \, [\exp(-k_1^0 \, t_i)]\} \, \Delta R = - \sum_{i=1}^{n} \{t_i \, (L_a^0)^2 \, [\exp(-k_1^0 \, t_i)] \\ &- t_i \, (L_a^0)^2 \, [\exp(-2 \, k_1^0 \, t_i)] - \sum_{i=1}^{n} \{R^0 \, t_i^2 \, L_a^0 \, [\exp(-k_1^0 \, t_i)]\}\end{aligned}$$

$$+ \sum_{i=1}^{n} \{y_i\, t_i\, L_a^0 [\exp(-k_1^0\, t_i)]\} \tag{22}$$

and

$$\sum_{i=1}^{n} \{t_i - t_i [\exp(-k_1^0\, t_i)]\} \Delta L_a + \sum_{i=1}^{n} \{t_i^2\, L_a^0 [\exp(-k_1^0\, t_i)]\} \Delta k_1$$
$$+ \sum_{i=1}^{n} t_i^2\, \Delta R = - \sum_{i=1}^{n} \{L_a^0\, t_i - L_a^0\, t_i [\exp(-k_1^0\, t_i)]\}$$
$$- \sum_{i=1}^{n} R^0 t_i^2 + \sum_{i=1}^{n} y_i\, t_i \tag{23}$$

Equations (21), (22), and (23) can be solved simultaneously for ΔL_α, Δk_1, and ΔR. New values for L_α^0, k_1^0, and R^0 may be determined from

$$\Delta L_a = L_a - L_a^0 \tag{24}$$
$$\Delta k_1 = k_1 - k_1^0 \tag{25}$$

and

$$\Delta R = R - R^0 \tag{26}$$

These new values may be substituted into Equations (21), (22), and (23), and ΔL_α, Δk_1, and ΔR recalculated. This process is repeated until ΔL_α, Δk_1, and ΔR equal some arbitrary small value. The final values for L_α^0, k_1^0, and R^0 yield the desired best fit of the model to the experimental data.

To solve Equation (5), the same procedure as above is followed, with the term $(R - P)$ replacing R in each equation.

To solve for the term k_2 in Equation (2), with the terms k_1, L_α, and $(R - P)$ known from the above calculations, Equation (2) is rewritten

$$D_i = \frac{k_1\, L_a}{k_2 - k_1} [\exp(-k_1\, t_i) - \exp(-k_2\, t_i)]$$
$$+ \frac{(R - P)}{k_2} [1 - \exp(-k_2\, t_i)] + D_a [\exp(-k_2\, t_i)] \tag{27}$$

At every time point, t_i, there is a corresponding D_i. Rewriting Equation (27) yields

$$D_i - \left[\frac{k_1\, L_a}{k_2 - k_1}\right] [\exp(-k_1\, t_i)] - \frac{(R - P)}{k_2} =$$
$$- \left[\frac{k_1\, L_a}{k_2 - k_1}\right] [\exp(-k_2\, t_i)] - \left[\frac{(R - P)}{k_2}\right] [\exp(-k_2\, t_i)]$$
$$+ D_a [\exp(-k_2\, t_i)] \tag{28}$$

or $$\left\{\frac{D_i - \left[\frac{k_1 L_a}{k_2 - k_1}\right][\exp(-k_1 t_i)] - \frac{(R - P)}{k_2}}{D_a - \left[\frac{k_1 L_a}{k_2 - k_1}\right] - \frac{(R - P)}{k_2}}\right\} = \exp(-k_2 t_i) \tag{29}$$

Then

$$ln\left\{\frac{D_i - \left[\frac{k_1 L_a}{k_2 - k_1}\right][\exp(-k_1 t_i)] - \frac{(R - P)}{k_2}}{D_a - \left[\frac{k_1 L_a}{k_2 - k_1}\right] - \frac{(R - P)}{k_2}}\right\} = - k_2 t_i \tag{30}$$

and $$\frac{1}{t_i}\, ln\left\{\frac{D_i - \left[\frac{k_1 L_a}{k_2 - k_1}\right][\exp(-k_1 t_i)] - \frac{(R - P)}{k_2}}{D_a - \left[\frac{k_1 L_a}{k_2 - k_1}\right] - \frac{(R - P)}{k_2}}\right\} = - k_2 \tag{31}$$

The solution involves putting the known values of k_1, L_α, and $(R - P)$, an assumed value for k_2^1, and one (t_i, D_i) pair into the left side of Equation (31) and calculating a new k_2, k_2^2. The above procedure is repeated with k_2^2 and k_2^3 calculated. This is continued until $k_2^n = k_2^{n+1}$. This procedure is repeated for each (t_i, D_i) pair and the average k_2 taken as the best value.

The values for the constants k_1, L_α, and R were calculated from the ''2-place'' data in column 2 of Table 2. The values were 0.10, 2.83, and 0.11, respectively. These same three constants were 0.10, 3.00, and 0.10 when the ''5-place'' data in column 2 of Table 2 were used. This illustrates the sensitivity of the equations to rounding and truncation.

Table 3 summarizes the solution of Equations (2), (4), and (5), both with the ''2-place'' and ''5-place'' data. The computer solution required 100 iterations for Equations (4) and (5) and 500 iterations for Equation (2). To demonstrate the adequacy of the model, Equation (2), and the method of solution, deficit data were generated using the constants $k_1 = 0.10$, $k_2 = 0.05$, $(R-P) = -0.19$, and $L_\alpha = 2.83$, the ''2-place'' solution in Equation (2).

Table 3. COMPARISON OF SOLUTIONS FOR CONSTANTS IN EQUATIONS (2), (4), AND (5)

	k_1		k_2		R		(R-P)		L_a	
	2-place data	5-place data	2-place data	5-place data	2-place data	5-place data	2-place data	5-place data	2-place data	5-place data
Assumed	0.10	0.10	0.05	0.05	0.10	0.10	−0.20	−0.20	3.00	3.00
Dark probe	0.10	0.10	-	-	0.11	0.10	-	-	2.83	3.00
Light probe	0.10	0.10	-	-	-	-	−0.19	−0.20	2.83	3.00
Unconfined probe	-	-	0.05	0.05	-	-	-	-	-	-

These data are plotted in Figure 4 and are compared to the deficit data in column 4 of Table 2. Note the expanded ordinate scale. The maximum deviation is -0.005 milligram per liter, which shows the usefulness of this computation method.

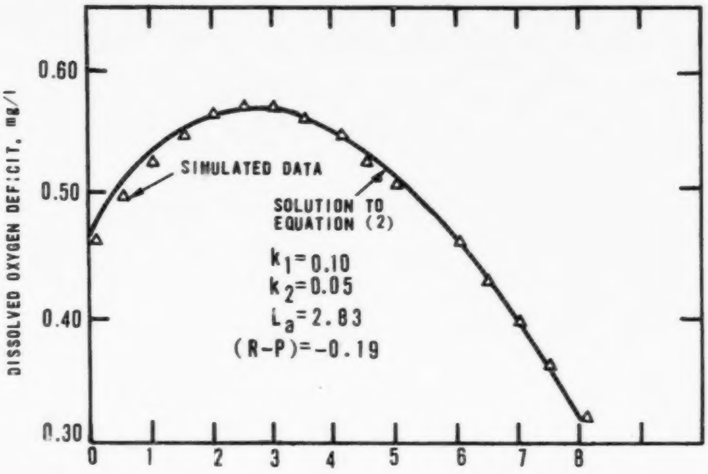

Figure 4. Comparison of the solution of Equation (2) with simulated data.

Discussion of "MANAGEMENT AND MEASUREMENT OF DO IN IMPOUNDMENTS"

Edward L. Thackston[*]

Although Equation (2) is a useful mathematical model of the DO budget in impoundments, the method presented for determining the parameters from measured data is inadequate. There are two principal criteria for acceptability and usefulness of a mathematical representation of a real system. First, the response predicted by the model must match closely the response of the real system. Second, the parameters determined by the model must bear a consistent, realistic relation to the processes influencing the real system. The authors have acknowledged the first criterion, but have ignored the second.

If the proposed model and method of solution is to be used for "studying the DO budget in impoundments," then the values of a parameter determined from data on several impoundments should be identical if the process represented by the parameter has the same influence on each. The only real reason for studying the DO budget is to determine what mechanisms are influencing it and what the relative magnitudes are. The proposed method of solution is obviously incapable of supplying estimates of parameters that will satisfy these requirements. The parameters resulting from solutions of the model by the proposed method using two sets of data cannot be compared, because there is no assurance that the estimated values will bear the same proportionality to the real values in both cases.

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The authors imply that the only criterion for acceptability of a solution to the model is whether or not the response predicted by the model conforms to the measured data. If this were true, a simple empirical polynomial could be fitted to the data and would be just as acceptable and probably easier to solve. If the only result desired is a curve fit to the measured data, and realistic values of the parameters are not needed, a line drawn through the points with a French curve would be more accurate, and the time spent in solving the mathematical model would be saved.

The writer tried solving the model using the algorithm proposed by Marquardt[12] and described by Thackston, Hays, and Krenkel.[13] The results were somewhat better than those obtained by using the authors' method, but errors were common. The values estimated with the model were quite sensitive to the initial guesses used. Bad fits or false fits could eventually be eliminated by comparing the standard errors of the different sets of estimates and selecting the lowest one, but this is still a slow and inefficient process.

As found by the authors, correct estimates of parameters could be obtained if the data were tabulated to six significant figures, but this is somewhat unrealistic.

The erratic behavior of the model and its insensitivity to varying values of the parameters can be explained by examination of Table 4. This Table shows a correlation matrix for the parameters L_a, k_1, and R, obtained in a fit to Equation (4), which is the first equation fit and thus the key to the model solution.

Table 4 shows that the parameters in Equation (4) are so highly correlated with each other that an error or misestimation in any parameter can be compensated for by a change in the other parameters, and the fit to the data will suffer little. Thus, small inaccuracies in data (even rounding to the nearest 0.01 milligram per liter,can lead to large errors in parameter estimation.

Further experimentation showed that the values of the remaining parameters could be estimated correctly to within 0.1 percent if the value of L_a were known beforehand. This is fortunate because this is a parameter that can be easily determined independently. A standard BOD test can be run on the impoundment water after it has been pasteurized to kill the algae and reseeded to provide a microbial population. The automatic-mixing DO probe can be used if desired. The data for oxygen depletion and time can easily be fitted, by the Marquardt algorithm to give L_a and k_1.

Table 4. TYPICAL CORRELATION MATRIX AMONG PARAMETERS IN EQUATION (4)

	L_a	k_1	R
L_a	1.000000	-0.994812	-0.996548
k_1	-0.994812	1.000000	0.983025
R	-0.996548	0.983025	1.000000

This is not the same k_1 that would be obtained in the impoundment environment, however, due to the different microbial populations; but the value of L_a should be accurate, because the total oxidizable carbonaceous organic content should remain the same.

The addition of this one extra step and the use of the Marquardt parameter estimation procedure will make Equation (2) a truly useful model for studying the DO budget in impoundments. The values of the parameters estimated from it should be accurate representations of the processes involved, and they should be comparable from situation to situation.

Discussion of

"MANAGEMENT AND MEASUREMENT OF DO IN IMPOUNDMENTS"

David A. Bella[*]

The writer disagrees that Equation (2) is a satisfactory mathematical model for representing the DO budget in impounded waters. The writer is somewhat confused by the authors' treatment of the coefficients involved in the mathematical model. (The term, ''coefficient,'' is used rather than the authors' ''constant,'' because they are seldom constant). The statement, ''The important feature of the model as used here is not the physical interpretation of the constants, but its ability to accurately reproduce what is occurring in nature,'' is cavalier. True, the worth of any mathematical model lies in its ability to describe what is occurring in nature. This should not be confused with the sole ability to be fitted to existing data. A simple Fourier series would work quite well. The point to be made is that a mathematical model is most needed, as mentioned by the authors, when (1) one wishes to roughly describe a condition for which exhaustive data are not available, and (2) one wishes to study the interrelationship of several factors. To perform either of the two above objectives requires a physical interpretation of the coefficients involved in the mathematical model.

Regarding the physical interpretation of Equations (1) and (2), it should be pointed out that their use in Reference 1 was not for a stratified impoundment but for a flowing stream in which complete vertical mixing was a justifiable assumption. Indeed, the concept of the reaeration coefficient, k_2, is based on the assumption of complete vertical mixing below the surface. The use of Equation (2) in a well-mixed body of water might be reasonable. The oxygen profiles taken by the authors in Boltz Lake, Kentucky, however, show DO variations from near saturation at the surface to zero or near zero at the bottom. One might wish to avoid this conflict by considering Equation (2) to be near valid in the euphotic zone only. If this were so, Equation (2) might then be only roughly used to describe the oxygen deficit in the one region of the impoundment where it is not a problem.

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The writer agrees with the authors to the point that any mathematical model, such as Equation (2), is a vast simplification of the physical situation, even in a mixed body of water. Engineering progress, however, has been made by working with the possible, admittedly often at the sacrifice of the ideal.

The one-dimensional stream is a case in point. The engineer realizes that variations occur with the depth and width of the stream. He also realizes that, for certain cases, these variations are far less pronounced than the variations along the length of the stream and the variations with time. He then develops a mathematical model that includes only the major variations with length and time. In doing so, he departs from the ideal and approaches the possible. Quite often this departure from reality is more a blind flight into nebulosity. Often, however, the one-dimensional approach, when judiciously applied, does represent a reasonable description of the physical situation, as in Reference 1. The greatest danger usually comes not in making the mathematical assumptions, but in believing them.

This one-dimensional reasoning might be applied to impounded waters. It could be reasoned that the horizontal variations in DO are small when compared to the vertical variations and the variations that occur with time. One might develop a mathematical model that includes only the major variations with depth and time. This one-dimensional vertical approach is by no means new, particularly in the study of the ocean.

Consider an impoundment slice of horizontal area, A, and of depth dz (Figure 5). Consider the depth, z, to have a value of zero at the bottom and a value of H at the surface. A positive water velocity is taken as movement from bottom to top. Performing a mass balance on such a slice might lead to the differential equation

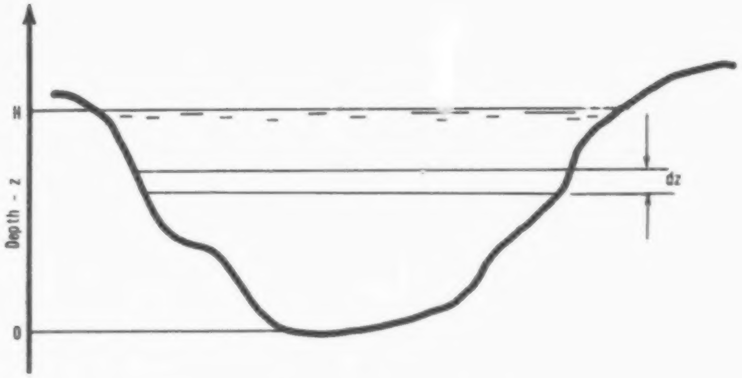

Figure 5. Impoundment.

$$\frac{\partial(AC)}{\partial T} = \frac{\partial(DA\,\partial C/\partial z)}{\partial z} - \frac{\partial(UAC)}{\partial z} - C_\alpha' - q_e C + q_i C_i$$

$$-k_1 AL + (P - R)A \tag{32}$$

As the horizontal area, A, at any depth may be considered constant with time, Equation (32) can be reduced to

$$\frac{\partial C}{\partial T} = \frac{1}{A}\left[\frac{\partial(DA\,\partial C/\partial z)}{\partial z} - \frac{\partial(UAC)}{\partial z} - C_\alpha' - q_e C + q_i C_i\right]$$

$$- k_1 L + (P - R) \tag{33}$$

in which

C = the dissolved oxygen concentration,

U = the upward water velocity averaged over a horizontal section,

D = the eddy diffusion coefficient in the z direction as well as the dispersion coefficient due to the velocity, U, variations across a horizontal section, similar to the longitudinal dispersion coefficient in a one-dimensional stream,

P = the rate of DO added per unit volume at a depth z by photosynthesis,

R = the rate of DO removed per unit volume at a depth z due to algal respiration,

L = the BOD concentration,

k_1 = the BOD decay coefficient, base e,

q_e = the flow rate of water exported per unit depth at a depth z,

q_i = the flow rate of water imported per unit depth at a depth z,

C_i = the DO concentration in the inflowing water,

and C'_a = the rate of DO removal per unit depth at a depth z due to other sources and sinks (shoreline-bottom demands, demands of other reduced substances, etc.).

Coefficients included in the above model can be functions of depth and time.

The effect of atmospheric reaeration is included by taking the total flux of DO into the impoundment at the surface as $A_0K_L(C_s - C_0)$

in which

A_0 = the impoundment surface area,

C_0 = the DO concentration immediately below the surface,

C_s = the concentration of DO at saturation,

and K_L = the liquid film coefficient for dissolved oxygen.

Finite-difference approximations to the above mathematical model, or similar models, are possible by several methods.

This mathematical approach might better fulfill the authors' stated objectives concerning a mathematical model that would permit general conclusions, predictions, and comparisons of one impoundment with another. A further objective of including the effects of destratification procedures on the DO resources might be facilitated. In the above example, pumping circulation might be included with a positive value of q_e in the lower layers and an equal value of q_i in the upper layers. The conservation of water would require a negative value of U. Higher values of D would be expected because of the increased turbulence and the variation of U across a horizontal section. Similar rough one-dimensional models might be developed for temperature, BOD, and other water quality parameters. It is hoped that these suggestions aid the quantitative description of their results.

Discussion of

"MANAGEMENT AND MEASUREMENT OF DO IN IMPOUNDMENTS"

C. H. J. Hull[*]

(Author's Note: Because the error in the solution to the mathematical model, originally pointed out by Hull in his Discussion, was corrected in Paper No. 4 before reprinting, the discussion that follows has been shortened somewhat.)

THE MATHEMATICAL MODEL

In this discussion, the writer follows the convention of using K_1 and K_2, instead of the authors' k_1 and k_2, as symbols for the BOD and atmospheric reaeration coefficients, respectively, where they appear in the differential forms of the various oxygen-balance equations, and where they appear as exponents to the base e in the integrated forms of the equations. The lower-case symbols are used, in this convention, as exponents to the base 10. This convention has served to distinguish between the coefficients used with the base e and those used with the base 10.

Before discussing the model for the dissolved oxygen (DO) budget in an impoundment, the writer would like to correct a misinterpretation by the authors of an earlier discussion by this writer. The authors stated that "...excessive concern over the precise meaning of these constants has... been criticized by Hull[2] and Ettinger[3]

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...'' The writer does not recall having criticized any such concern for being excessive. On the contrary, this writer has long advocated greater concern over the meaning of these so-called constants. As evidence of such concern, the writer is reluctant even to call K_1, K_2, L_α, P, and R constants. Although they may be considered as constants in a mathematical equation representing a given condition, they are generally not constant in nature. The authors recognized this, as shown by their reference to ''...the changes in constants, as related to changing sunlight, wind, algal counts, and nutrient level...'' In view of the inconsistent nature of K_1, K_2, L_α, P, and R, they are better called coefficients, values, factors, or parameters. If concern over their meaning is a fault, then the writer must place himself among the culpable, not among those who have criticized the concern.

What this writer has criticized is the excessive work in the usual method of determining K_1, the BOD coefficient, for streams–work that serves only to introduce errors. Criticism of excessive, error-introducing work that results in a meaningless value of K_1 should not be interpreted as criticism of concern over the meaning of the coefficient. The excessive work referred to is the sampling, transporting, cooling, storing, neutralizing, diluting, seeding, and incubating stream water at a standard temperature, the method usually employed to determine a laboratory K_1, and the mathematical conversion of this value to a stream K_1 by means of a temperature-correction formula of no validity.[14] Instead of the involved, error-introducing procedure described above, the writer has advocated incubation of undiluted, untreated BOD samples in dark bottles in the stream itself to eliminate delay, temperature changes, and other complications inherent in the laboratory technique. The authors' procedure, involving *in situ* incubation, is in accord with the writer's preferred practice in this respect.

The authors attribute to the writer a suggestion that all DO sources and sinks in a given impoundment be measured to evaluate the DO budget; the writer has made no such suggestion. The authors gave no literature reference for this alleged suggestion, but are probably referring to the writer's discussion[2] of the paper by O'Connell and Thomas.[1] In that discussion of the application of a mathematical model (similar to Equation (2)) to short reaches of a shallow stream, it was suggested that this model offered little advantage over a simple algebraic summation of DO sources and sinks for each short reach. The suggestion, when made, was not intended to apply to an impoundment.

The authors are correct in citing the advantages that would accrue from development of a reliable model. It does not follow, however, that an algebraic-summation model would be any less reliable than the one they use. They suggest that the simple-summation method is time-consuming and evaluates the DO budget under only one set of conditions, whereas the reliable model would provide for changes in the so-called constants, permitting general conclusions to be drawn and predictions to be made, and allowing comparisons among impoundments. It is not at all clear why the authors assume, apparently, that the values obtained in the simple-summation method cannot be corrected mathematically for changed environmental conditions just as easily as the unconstant constants in Equation (2). After all, Equation (2) is merely the integrated form of Equation (1), which is nothing more than a simple-summation of oxygen sources and sinks.

It should be noted that the authors do not apply their procedure to an entire impoundment, but only to the euphotic zone. As discussed later, it is probably necessary to limit the application of Equation (2) to a well-mixed layer that is usually shallower than the euphotic zone.

A mathematical model for DO changes, similar to one proposed in 1965 for a shallow stream by O'Connell and Thomas,[1] is used by the authors in their paper to apply to the euphotic zone of impoundments, where, they say, sedimentation is not an important factor. Presumably, they refer to sedimentation as it relates to the removal of BOD from the euphotic zone, or to the oxygen demand of bottom deposits. If so, it appears valid to exclude this demand from a model representing the euphotic zone for relatively short periods. However, it should be recognized that sedimentation (settling) does occur in the upper layers of reservoirs, and this can have an indirect but significant long-term effect on the oxygen balance of the euphotic zone. For the relatively short time periods (8 hours) considered by the authors, sedimentation can probably be ignored, except where settleable solids are obviously present, e.g., near the head of an impoundment fed by a turbid stream. For even shorter periods of analysis, as recommended further on in this discussion for other reasons, the direct effects of sedimentation and bottom deposits can be ignored even more safely. The effects, however, of vertical density currents that, according to Bradley,[15] carry DO and particles from near-surface layers downward into the hypolimnion, may require consideration and possibly modification of the mathematical model or procedures of sampling oxygen concentrations.

One indirect effect of sedimentation on the oxygen balance of waters over bottom deposits of organic matter is that of ebullition of anaerobic fermentation gases from such deposits. These gases form bubbles, which rise through the overlying water and purge oxygen from the water (Hull).[16] Equation (2) would not account for this oxygen sink in an impoundment or area of an impoundment where and when ebullition takes place.

In shallow areas of impoundments, a well-mixed euphotic zone may extend from the surface to the bottom, where both benthic oxygen demand, benthic primary production, including that of rooted aquatic plants, and respiration of benthic autotrophs could be expected to influence the oxygen balance. For example, in 1967 Smith, Hall, and Stanley[17] described excessive growths of Eurasian water milfoil in reservoirs of the Tennessee Valley, thick enough to threaten navigation. It seems obvious that such growths must exert a significant influence on the overall oxygen balance of an impoundment. Moreover, it should not be inferred that visible macrophytes alone are significant; benthic microphytes and heterotrophs will generally have to be reckoned with in the total oxygen budget. Although these factors may not rule out the use of Equation (2) for determination of the atmospheric reaeration coefficient, K_2, or other parameters in the surface layer of a deep offshore area, they may well invalidate this mathematical model as a representation of the oxygen budget of the entire reservoir.

The model does not account for flux of DO from the euphotic zone downward into the deeper layers of water. Ignoring this factor in a model for the euphotic zone is valid only if the bottom of the euphotic zone is in a layer in which there is no vertical DO gradient. Studies by the authors on Boltz Lake, Kentucky show the vertical distribution of dissolved oxygen. The DO curves indicate no gradient from the surface to a depth of about 5 feet. Generally, one can expect the euphotic zone to extend to much greater depths in large reservoirs. This does not necessarily negate the use of the model; it means only that the model should be applied to a surface layer at the bottom of which there is no significant vertical DO gradient. This layer will not generally include the entire euphotic zone. In a 1957 study of Roanoke Rapids Reservoir on the Roanoke River in North Carolina, a mathematical model practically identical to Equation (1) was solved by Pritchard and Carpenter for a surface layer 1 meter deep, as reported in 1963 by Hull.[18] The 1 meter depth was selected partly to avoid the complexity of a downward flux of oxygen; it was observed that there was no consistent vertical oxygen gradient at a depth of 1 meter.

The authors rightly state reasons why a reliable mathematical model of the DO budget of impoundments is needed. Among other reasons, they explain that a reliable model would allow predictions to be made, taking into account changes in the constants as related to various environmental conditions. Later, they conclude that Equation (2) is a satisfactory mathematical model for representing the DO budget. This conclusion is apparently based on the fairly good agreement obtained between the smoothed observed DO data and the DO curve resulting from solution of Equation (2) using the smoothed data from the three probes for an 8-hour period.

It should be recognized that what the authors have done is merely fit a curve to observed data. This hardly constitutes proof that the mathematical model is reliable for predictions of DO concentrations under different conditions of wind, sunlight, temperature, algal counts, nutrient levels, turbidity, turbulence, and a host of other oxygen-controlling factors. There are probably many mathematical models that can be fitted to the observed DO data with equally good or better agreement, but this would not necessarily justify their use for predictions. This fact alone does not mean that Equation (2) is not adequate or reliable for such predictions. Rather, this argument should be taken as a suggestion for further testing of Equation (2) under conditions covering the range of oxygen-controlling factors expected in impoundments. Only after such testing—with positive results—will a claim of reliability for the model be justified, and then it will be useful for predictions only if methods are provided for predicting the ''constants'' under various conditions.

On the other hand, it should be pointed out that although the differential form of the mathematical model, Equation (1), is probably a valid short-term expression for the DO budget of a shallow, well-mixed surface layer within the eutrophic zone of an offshore area of an impoundment, the integrated form, Equation (2), is not valid for long-term predictions. An equation similar to Equation

(2) was derived by O'Connell and Thomas[1] by assuming that (*P–R*) is a constant, obviously an invalid assumption. In using Equation (2) in their Truckee River study, they attempted to overcome this weakness by applying the equation only to relatively short reaches of the stream involving short time-of-flow periods during which *P* and *R* would not be expected to vary widely. However, Symons et al. seem to have placed no such limitation on their use of the integrated form of the mathematical model. In applying Equation (2) to a period as long as 8 hours, the authors have surely gone beyond the limits of validity for the assumption of constant (*P–R*). It is well known that *P* varies widely during any daylight period of 8 hours. This is one of the many reasons, some of which are discussed below, for limiting the application of the authors' equipment, mathematical model, and procedures to the shortest period of time that will provide significant differences in DO concentrations in the light and dark containers. For accurate computation of K_2 by the simplified method presented later in this discussion, the test period should be at least 4 hours, however. The simplified method does not depend on an assumption that *P* or *R* is constant. This will allow use of the model for determining approximate–and reasonably reliable–values of the various so-called constants for the environmental conditions prevailing during the test period. The writer fails to see, however, how Equation (2) can be used to predict DO concentrations for more than an hour or so with any degree of reliability, even for a shallow surface layer of water in an offshore portion of an impoundment. For an entire impoundment, including the shallow littoral areas and the deeper portions of the trophogenic zone–as well as the hypolimnion, the model has more severe limitations as a prediction tool.

The authors' solution of the mathematical model resulted in curves of oxygen deficits over time that agree only approximately with the observed data, and this approximate agreement is shown for a period of only 8 hours. Although the differences between the computed deficits and observed or simulated deficits are quite small within the time periods considered, it should be noted that 8 hours is a short period in relation to the average flow-through period of many large reservoirs. The flow-through period in a typical large reservoir may be measured in weeks or months rather than hours. The authors have not shown the degree of differences between predicted and observed or simulated deficits for such longer periods.

Even for the 8-hour periods considered, the indicated small errors in the computed deficits may be the result of compensating errors in the computed values of K_1, K_2, L_α, P, and R. For example, in the problem based on observations of artificial impoundment water, the authors computed a value of 0.06 milligram per liter per hour for P. As will be shown later herein, the correct value of P in that problem is probably 0.01 milligram per liter per hour, and the indicated error in the authors' value is 500 percent. With an error of such magnitude, it does not appear valid to conclude, on the basis of close agreement between computed and observed or simulated deficits, that Equation (2) ''...is a satisfactory mathematical model for representing the DO budget in impounded water.''

For still another reason, it seems premature to conclude that Equation (2) is satisfactory. The authors have presented no evidence showing that the model represents the DO budget in impounded water. They used simulated DO deficits and deficits observed in ''artificial impoundment water'' to demonstrate a mathematical curve-fitting procedure. These demonstrations, even if they had been successful, would hardly constitute valid proof that Equation (2) could be used successfully to fit a curve to observed deficits in an actual impoundment, where they would have to deal with ''...changing sunlight, wind, algal counts, and nutrient level...'' during the period of observation. The curve-fitting demonstrations in the authors' paper are not valid proofs that Equation (2) can be used to predict DO concentrations in the natural impoundments beyond the period of observation.

It appears that a reliable model is still not available for the DO budget of an entire reservoir. Although the writer has disclaimed previous authorship of the suggestion that a simple-summation model be applied to impoundments, perhaps such a suggestion is now in order. The DO budget of an impoundment, for the general case, should account for such assets as advective gains (inflow), atmospheric reaeration, and photosynthesis by phytoplankton, benthic microphytes, and macrophytes. Debits in the oxygen budget generally include advective losses (outflow), respiration of heterotrophs and autotrophs (both suspended and benthic forms of each), and losses to the atmosphere as a result of supersaturated surface water, or as a result of ebullition of anoxic fermentation gases from bottom muds, gases that purge oxygen from the water. The values of the various constants of Equation (2), determined by short-term observations using the three-probe technique, would be useful in an algebraic-summation model accounting for all oxygen sources and sinks.

APPLICATION OF THE MODEL

It should be noted that the DO probes developed by the authors are really modifications of the simple bottles used in the traditional light and dark oxygen-difference method. To show the variation of photosynthesis and respiration (including BOD and phytoplankton respiration) with depth, it would be necessary to expose simultaneously a series of the special DO-probe containers at various depths throughout the eutrophic zone, just as it would be necessary with simple light and dark bottles.

To be meaningful, the values of the parameters D, K_1, L_a, P, R, and y appearing in the model should be the mean values for the layer of water considered. It appears that the authors have applied the mathematical model to data from a single set of the special probes exposed at a single depth. If so, it is pertinent to ask what the exposure depth should be, how it is to be selected, and how the values of these parameters obtained for that depth relate to the mean values of the parameters for the euphotic zone. It is not clear to the writer how single-depth measurements can be used directly to solve a model representing the DO budget of the euphotic zone. (Actually, the model is valid only for a shallow layer, the bottom of which has no vertical DO-concentration gradient; but, even in such a layer, photosynthetic oxygen production rates would vary with depth.)

If the authors have indeed used single-depth observations, their procedure needs modification to obtain mean values of D for all three conditions (dark confined water, lighted confined water, and lighted unconfined water) for the range of depth of the surface layer of water to which the mathematical model is to be applied.

LIGHT AND DARK SAMPLE CONTAINERS

The authors have documented their awareness of many of the potential inaccuracies of the light- and dark-bottle techniques of measuring photosynthetic oxygen production. They suggest that the special equipment they have used in their DO-probe method minimizes these inaccuracies. Does it really? Let us examine some of the known or suspected bottle errors and see whether the authors' equipment reduces or minimizes these errors.

One of the commonly suspected bottle errors is that related to the growth of periphytes on the inside surface of the container. These surface organisms utilize oxygen, just as do the free-suspended organisms distributed throughout the sample in the bottle. Thus, the oxygen consumption rate in the bottle is probably faster than in an unconfined parcel of the same water of equal volume, at least after a finite incubation period during which the periphytic population increases to a significant level. This is one of the reasons for the frequently used procedure of limiting the period of light- and dark-bottle tests to the shortest period that will produce a significant change in DO concentration. The authors have indicated incubation periods of as long as 8 hours (Figure 1) in their studies. If any significant error attributable to periphytes is involved, it could be reduced by shortening the period of exposure of the three probes. This type of improvement–one of procedure, not equipment–can be applied to bottles no less than to the authors' special containers.

Another method of reducing any periphytic effect is to use containers with a relatively small surface-to-volume ratio. Symons[9] has elsewhere described the special sample containers used by the authors as having a surface-to-volume ratio ''...nearly one-third that of a standard 300-ml bottle –,'' presumably a standard BOD bottle. It follows that the authors' special containers should cause a lesser periphytic effect than a 300-ml bottle. There is no reason, however, to limit the size of bottles used in light- and dark-bottle tests to 300-ml. Merely by selecting large bottles, with a surface-to-volume ratio smaller than that of the authors' special containers, one could further reduce any periphyte-related error. Of course, assuming that the special containers are desirable because of their other features, they also could be made larger to reduce the surface effect. For the size described by Symons,[9] however, the authors' claim of minimized inaccuracies of the type attributed to the inside surface of the bottles appears unjustified. The surface error relative to a small bottle is reduced, but hardly minimized.

It should be noted that if periphytic heterotrophs are favored in containers relative to unconfined waters, they should be favored approximately equally in both light and dark containers of the same size and shape.

Thus a higher heterotrophic respiration rate in the dark container would be approximately equaled by a higher heterotrophic respiration rate in the light container. When the light- and dark-bottle test is used to determine gross photosynthesis, *P*, the dark bottles serve mainly to correct for respiration in the light bottles, and therefore one can expect this error to be largely compensated. In 1959, Pratt and Berkson[19] presented data supporting this interpretation, showing no differential bacterial growth rates in light and dark bottles during reasonably-long exposure periods (46 hours).

This compensating effect applies only to gross production. For net production, (*P–R*), as traditionally determined in the light- and dark-bottle technique, only the light bottle is used, and thus any error due to heterotrophic periphytes on the inside surface of the bottle is not compensated.

There is, of course, a possibility that periphytic autotrophs are favored by the surface effect in the light container, bottle or otherwise. Also, as suggested by Pratt and Berkson,[19] it is possible that the surface effect may indirectly enhance the environment for both periphytic and free-suspended forms of algae in the light bottle because of the demineralization of organic matter by the surface-enhanced periphytic heterotrophs. This effect, termed an auto-accelerating process by Goldman,[20] is one of the unavoidable sources of error in light and dark methods of measuring primary production, whether measured in bottles or special containers such as used by the authors.

These effects suggest that the containers should be as large as practical and the period of exposure no longer than necessary to show a significant change in DO content. The authors could easily further reduce the potential inaccuracies of their light and dark oxygen method in these two ways. Shortening the exposure period alone would improve the reliability of their results, with no cost for modifying equipment, and with a saving of costs related to field work and solution of the mathematical model.

Another bottle error the authors claim to have minimized with their equipment is that associated with the difference–presumed to be significant–between turbulence in bottles and turbulence in the unconfined water, for which the various so-called constants are sought. It is to their credit that they have recognized this potential error and have attempted to minimize it. It is not at all certain, however, that this error is significant in all situations, nor is it certain that where significant, the turbulence error is minimized–or even reduced–by means of their special equipment. In fact, it appears quite probable that the special equipment used by the authors provides turbulence that is atypically high for many waterways, especially impoundments.

In 1961 laboratory experiments, discussed briefly elsewhere by Hull,[21] gross photosynthetic oxygenation rates, *P*, in BOD bottles were found to be a continuous function of stirring rates between a rate of zero (quiescent condition) and a stirring rate in excess of 400 rpm. In these experiments, *P* increased gradually with the stirring rate up to a maximum of about 15 percent above the quiescent value of *P*.

The purpose of these laboratory tests was to evaluate the bottle error related to turbulence in a study of photosynthesis in the tidal Delaware River. In addition to the laboratory tests, field observations were made of turbulence in the unconfined river water and in a bottled sample of river water suspended in the river in the manner usually employed in the light and dark measurements of photosynthesis. In these field observations, the response of a DO electrode (similar to that used by the authors) was taken as a measure of small-scale turbulence for a given oxygen concentration in both the unconfined and bottled river water. Similar electrode observations were made in the laboratory to relate turbulence to stirring speeds and photosynthesis rates. Based on these laboratory and field tests, the writer concluded that the small-scale turbulence in the bottles suspended in the river was greater than the turbulence in the unconfined river water, and that the resulting error in the bottle *P* obtained was an increase of about 3 percent. The increase in bottle turbulence and *P* was attributed to the agitation of the suspended bottle by the tidal current, which is quite swift in the Delaware River. A parcel of unconfined water carried along with the current is not agitated nearly as much. Although the unconfined water is subject to large-scale eddies that are eliminated from the bottled samples, it is the small-scale diffusion mechanisms, including molecular diffusion and very small-scale eddy diffusion that are involved in turbulence effects on short-term photosynthesis rates.

It should not be inferred that the maximum turbulence-related increase of only 15 percent observed for Delaware River water would apply to less nutrient-enriched waters such as would be expected in most impoundments. The influence of turbulence is probably dependent, in an inverse relationship, upon the concentration of nutrients in a given water. Thus, in slightly enriched waters, the effect of turbulence would probably be much greater, and the potential error in containers providing atypical turbulence would be correspondingly greater. It follows that for some-probably most-impoundment water, it would be possible to vary artificial turbulence to obtain an increase of considerably more than 15 percent in photosynthesis rates over the rates for quiescent waters in containers, and probably even over the rates in unconfined quiescent waters.

The point of these observations is that if turbulence is a factor necessitating correction for confined samples of water, the correction applied should be in the right amount and in the right direction. From the writer's experience, he is led to believe that the artificial turbulence induced by the propellers in the authors' special containers would produce small-scale turbulence that is more intensive than the small-scale turbulence prevailing in most impoundments under most conditions. Conceivably, under certain weather or current conditions in impoundments, the samples in the containers, even in the absence of the propellers, could be agitated more than a parcel of unconfined water at the same depth, as was observed in the tidal Delaware River, in which case the correction by the propellers would increase the error; that is, the correction would be in the wrong direction.

The propeller speed in the special containers is probably designed primarily to circulate water past the DO probes. If the minimum propeller speed required for this purpose is less than that required to equalize the inside and outside effects of turbulence on photosynthesis and respiration, then it may be possible to design a variable speed propeller that could be adjusted as necessary to compensate for the reduction of turbulence in the container.

On the other hand, in the more likely event that the minimum propeller speed adequate for the DO probe is greater than required to equalize the inside and outside turbulence effects on photosynthesis, the two purposes of the propeller could not be reconciled at the same time. In this case, three error-reducing methods occur to the writer: (1) Reducing the speed of the propeller (or stopping it), except when DO readings are necessary; (2) using a larger container to reduce the overall effect of the propeller speed necessary for use of the DO probe;

or (3) evaluating the error as was done in the Delaware River study, cited above, and correcting for it after the fact.

All three methods require some measure of small-scale turbulence inside and outside the containers. Method (1) would tend to negate one of the advantages claimed for the DO probes, the ability to provide continuous readings. Continuous readings are not necessary, however, for solution of the mathematical model used by Symons et al. In fact, their data indicate that only periodic spot readings, from which the authors obtained smoothed DO curves, were taken. Symons[9] has explained elsewhere that this procedure was used to extend the life of the batteries that powered the DO probes. Method (2) would require a number of containers of different sizes for use in different waterways. Optimum container size for turbulence equalization in a given waterway might be smaller, however, than the size required to minimize the surface effect discussed above. Method (3) would allow use of a given set of the light and dark containers for study of any waterway.

Taking everything into account, it would probably be advisable, for general oxygen-balance studies, to use a single set of special containers of the type designed by the authors, making them as large as possible to minimize the surface effect. The addition of some means of varying agitation would allow control of turbulence above the level provided by the DO-probe propeller. Once the size was standardized, it would be possible to study the relationships among such factors as nutrient levels, BOD levels, turbulence, respiration, and primary production in the containers. Then the error related to turbulence could be reduced by increasing or decreasing agitation within the limits of the control device. Any residual error could be determined as was done in the Delaware River study and used to correct the results obtained.

One factor that has been suspected as a source of error in bottle measurements of primary production is the light-transmission properties of the glass of which the bottles are made. It is therefore pertinent to ask whether the material used in construction of the special containers has light-transmission properties better or worse than glass (when both are compared to the water being tested). In other words, is the light reaching a sample enclosed in the special light container representative of light illuminating a parcel of unconfined water at the same depth? Is it more or less representative than light reaching a sample in a glass bottle? This question needs to be answered before a claim of reduced or minimized inaccuracies is justified.

Apart from the light-transmission properties of the walls of the special containers used by the authors, there is undoubtedly some shadow effect, probably small, of the opaque materials making up the DO probe and propeller. This effect could be evaluated by chemical DO analysis of eutrophic water samples after exposure in two of the light containers, one with and one without an opaque probe assembly. If the error was found to be significant, a correction could be applied to regular test results from the light probe. Also, any such error could probably be reduced by enlarging the containers—another reason for making them as large as practical.

It should be clear from the foregoing that the special equipment used by the authors does not necessarily minimize—nor does it necessarily even reduce—the inaccuracies of the bottle-oxygen method of measuring photosynthesis. This is not a crippling criticism of the general design, however. Even if the accuracy is no better than that of bottles (the accuracy can be improved by enlarging the design, as discussed above) some of the other advantages appear to warrant further development and use of the equipment. For example, the equipment can be used for short-period daytime field studies without exposing samples to atypical light, as explained by the authors. Having carried out a great deal of field work between sunset and sunrise (in a busy port at great inconvenience and risk) to avoid sample exposure to atypical light during daylight hours, this writer can readily appreciate this feature of the special containers. For investigators expecting to perform many light and dark measurements, the time and expense necessary to acquire and perfect special containers seem justifiable.

On the other hand, investigators of stream oxygen relationships who have only infrequent need to measure primary production may not be able to justify the investment of time and money for special light and dark containers such as those used by the authors. The occasional investigator can use the traditional bottle technique to learn a great deal about primary production and respiration of the planktonic community in a waterway. By using large bottles, by avoiding exposure of deep-water samples to direct sunlight, and by accepting some added work load in sampling and analysis, he can obtain results that are inherently more representative of the stream than the traditional BOD test performed in the laboratory. (It has long puzzled the writer that stream analysts who are quick to criticize the reliability of *in situ* bottle measurements of photosynthesis appear to have unquestioning faith in laboratory bottle measurements of BOD. The latter depart much further than the former from natural conditions in the unconfined waters of a stream.)

SOLUTION OF THE MODEL

The authors have presented several pages of complex mathematical formulations in their Appendix to illustrate their solution of the model for the problem represented by the data in Table 2. Their solution method requires the use of a high-speed electronic computer. The time, effort, and expense required to program the computer input must have been considerable.

It seems to this writer that the authors have made a rather simple problem appear extremely difficult. Any alert investigator familiar with the oxygen balance of surface waterways and with the light and dark oxygen method of measuring photosynthesis should be able to inspect the data in Table 2, and state immediately the correct value of *P*. This can be done without even pencil and paper, let alone a high-speed electronic computer. From the inspection of the deficits in Table 2, it can be seen that *P* must be 0.30 milligram per liter per hour. The solution by inspection is explained below.

Solution for *P*

As pointed out by Hull in 1950[22] and again in 1964,[21] the gross oxygen production by photosynthesis in the light containers is equal to the difference between the final concentration in the dark container, provided the initial concentrations in the two containers are the same, or

$$\int_{t=o}^{t=t} P\,dt = (\text{Final DO})_{\text{light}} - (\text{Final DO})_{\text{dark}} \qquad (34)$$

As the authors have assumed that *P* is constant, it follows (for this simulated problem) that

$$Pt = D_{\text{final, dark}} - D_{\text{final, light}} \qquad (35)$$

From the authors' data presented in Table 2, for time of 8.0 hours

$$Pt = 2.92 - 0.52 = 2.40 \text{ mg/l}$$

$$P = \frac{2.40 \text{ mg/l}}{8.0 \text{ hour}} = 0.30 \text{ mg/l per hour.}$$

This simple solution for P can be carried out with mental arithmetic-and it provides the correct answer.

Of course, it should not be assumed that P will be a constant in an actual waterway. However, Equation (34) or (35) will always provide an average value of P for the period of exposure. By making a series of short-term light and dark tests, average values of P can be obtained by this simple method that will approach the instantaneous rates of photosynthetic oxygen liberation. Moreover, short exposure periods will tend to minimize the surface effects of the light and dark containers, as discussed earlier.

Solution for K_2

The writer has developed a simplified solution method for K_2 that can be applied to a more general case in which the DO concentrations (or deficits) in the confined and unconfined lighted waters differ significantly after a period of time. The following equation can be used to determine this coefficient directly, using observed DO deficits:

$$K_2 = \frac{6[(D_L - D_U)_t - (D_L - D_U)_\alpha]}{T[(D_U)_\alpha + 4(D_U)_{t/2} + (D_U)_t]} \tag{36}$$

in which

K_2 = the atmospheric reaeration coefficient, hour^{-1},

D_L = the deficit in the confined, lighted water,

D_U = the deficit in the unconfined water,

α = subscript indicating initial conditions for the time period over which the equation is applied,

t = subscript indicating the final conditions for the time period over which the equation is applied,

$t/2$ = subscript indicating the half-time conditions; that is, midway between the times represented by subscripts a and t,

and T = elapsed time (the period between initial time a and final time t).

The derivation of Equation (36) will be explained in a paper now being prepared for publication in a future issue of the Journal, Sanitary Engineering Division, Proceedings of ASCE.

If Equation (36) is applied to the deficits for the entire 8-hour period represented in Table 2, the following substitution and solution result:

$$K_2 = \frac{6[(0.52 - 0.32) - (0.47 - 0.47)]}{8[0.47 + 4(0.55) + 0.32]} = 0.05017 \text{ hour}^{-1}$$

$$= 0.05 \text{ hour}^{-1}. \tag{37}$$

It is not necessary to use the entire 8-hour period. To illustrate, Equation (36) is applied to the 4-hour period between time 1.5 hours and time 5.5 hours, so that T equals 4 hours and the half-time deficit is that at time 3.5 hours in Table 2. The substitution of the deficits in the equation gives

$$K_2 = \frac{6[(0.64 - 0.49) - (0.59 - 0.55)]}{4[0.55 + 4(0.56) + 0.49]} = 0.05030 \text{ hour}^{-1}$$

$$= 0.05 \text{ hour}^{-1}. \tag{38}$$

These calculations, based on deficits correct to only three significant figures (two decimal places), give values of K_2 that, when rounded to three significant figures, agree exactly with the correct (assumed) value of K_2, 0.05 hour^{-1}. Thus, this simple paper-and-pencil method of solution compares favorably with the complex computer method presented by Symons et al.

A word of caution is in order here. Equation (36) has been used to obtain 64 independent solutions for K_2, including the two shown above. These solutions employed all combinations of deficits in Table 2 that were possible without interpolation between the half-hourly deficits. Of the 64 values of K_2 obtained, 47 agreed with the assumed value to the second decimal place. The 17 values that did not agree to the second decimal place resulted from application of Equation (36) to lapsed-time periods of only 1, 2, or 3 hours. For short time periods, the calculated value of K_2 is relatively sensitive to the rounding errors in the simulated deficits. For long time periods, however, these rounding errors influence the results to a lesser degree. For this reason, Equation (36) should be used for time periods of 4 hours or longer; that is, the value of T in the denominator should be at least 4.0 hours. (It should not be inferred that the simplified solution for P, using Equation (34) [or Equations (35) or (39), which follows], also requires a lapsed-time period of 4 hours or longer.)

Solution for K_1, L_a, and R

There are five unknown parameters in the authors' simulated problem. With the simplified methods explained above, accurate direct solutions are possible for two of these parameters, P and K_2. This leaves only three unknowns (K_1, L_a, and R) to be found-a much easier task than that faced by Symons et al. with their computerized method of solution.

SOLUTION OF OBSERVED LABORATORY PROBLEM, P AND K_2

The writer has applied simplified methods of determining P and K_2 to the problem represented by the observed deficits in artificial impoundment water presented by Symons et al. in their Table 1. In this case, Equation (35) is modified to account for the difference between the initial deficits in the light and dark containers. The modified equation for gross photosynthesis is

$$Pt = (D_{\text{final, dark}} - D_{\text{final, light}}) - (D_{\text{initial, dark}} - D_{\text{initial, light}}) \tag{39}$$

Substitution of the appropriate time and deficits from Table 1 in Equation (39) yields

$$P = \frac{(3.41 - 3.27) - (2.82 - 2.77)}{8}$$

$$P = 0.01125$$

$$= 0.01 \text{ mg/l per hour.} \tag{40}$$

Based on the accuracy of the simplified model for P, as demonstrated in the solutions of the simulated problems, it is assumed that the above value of P is the correct value. If so, it follows that the authors' value, 0.06 milligram per liter per hour, is much too high.

In solving for K_2 of the artificial impoundment water, Equation (36) can be used without modification.

$$K_2 = \frac{6[(3.27 - 2.98) - (2.77 - 2.74)]}{8[2.74 + 4(2.97) + 2.98]} = 0.01108$$

$$K_2 = 0.01 \text{ per hour} \tag{41}$$

This is in agreement with the authors' computer solution.

COMPARISON OF COMPUTATIONAL METHODS

Table 5 shows a comparison of the results of the complex computer method presented in the authors' paper and the simplified method presented in this discussion. Unfortunately, simplified solutions for K_1, R, and L_α have not been found as yet. The authors' values for these three parameters as well as for $(R-P)$ are included in the table for completeness.

Comparison of the absolute and percentage errors of K_2 and P obtained by the two computation methods shows the simplified method to be accurate. Thus, both simplicity and accuracy appear to dictate the use of the writer's methods of computing these two parameters.

The simplified, direct solutions of the mathematical model for K_2 and P should make it possible to decrease the complexity and increase the accuracy of the solutions for the other three unknowns, K_1, L_α, and R. The authors' method of solving the mathematical model is unnecessarily difficult and complex. It should probably be put into that class of research efforts called ''mathematical doodlements'' by Ettinger.[3]

SUMMARY

The authors appear to be on the track of a useful approach to understanding and control of the oxygen balance of impoundments. Considerable improvement seems possible, however, in the equipment and methods used in

Table 5. COMPARISON OF SOLUTIONS

Source of data	Parameter, units	Correct value	Complex method (Symons et al.) Value obtained	Complex method (Symons et al.) Error, percent	Simplified method (Hull) Value obtained	Simplified method (Hull) Error, percent
(1)	(2)	(3)	(4)	(5)	(6)	(7)
Simulated (Table 2)	K_1 (per hr)	0.10	0.10	0.00	--[b]	--[b]
	K_2 (per hr)	0.05	0.05	0.00	0.05	0.00
	P (mg/liter/hr)	0.30	0.30	0.00	0.30	0.00
	R (mg/liter/hr)	0.10	0.11	10.00	--[b]	--[b]
	R-P (mg/liter/hr)	-0.20	-0.19	5.00	--[b]	--[b]
	L_a (mg/liter)	3.00	2.83	5.66	--[b]	--[b]
Observed[a] (Table 1)	K_1 (per hr)	unknown	0.31	--[b]	--[b]	--[b]
	K_2 (per hr)	unknown	0.01	0.00[c]	0.01	0.00[c]
	P (mg/liter/hr)	unknown	0.06	500.00[c]	0.01	0.00[c]
	R (mg/liter/hr)	unknown	0.05	--[b]	--[b]	--[b]
	R-P (mg/liter/hr)	unknown	-0.01	--[b]	--[b]	--[b]
	L_a (mg/liter)	unknown	0.51	--[b]	--[b]	--[b]

[a] Deficits obtained by Symons et al. using their three-probe procedure in artificial impoundment water.

[b] Value not computed or not reported for method indicated.

[c] Because of the mathematical accuracy of the simplified method, as demonstrated in solutions of the simulated problem, the simplified-solution values for P and K_2 are assumed to be the correct values for the purpose of computing percentage errors.

obtaining field data, and in their methods for solving the mathematical model of the oxygen budget.

The proposed mathematical model is not a valid general expression for the oxygen balance of an entire reservoir; nor is it valid for the entire euphotic zone. The model presented should be used only to represent an offshore surface layer, generally shallower than the euphotic zone, across the bottom of which there is no significant flux of oxygen. Some of the parameters obtained by solution of the model for an offshore area of limited depth would be useful in a more comprehensive mathematical model for the entire reservoir.

The authors appear to have used single-depth measurements of DO deficits for solution of the model representing, they say, the euphotic zone. If so, they should modify their procedure to obtain the mean values of the deficits for the entire layer of water to which the mathematical model is applied.

Contrary to the authors' claim, the special sample containers used probably do not minimize the errors associated with the light- and dark-bottle technique of measuring photosynthesis. Whether the special containers used even reduce these errors is questionable. There are ways to reduce or correct these errors, however, and certain advantages of the special containers appear to warrant their further development and use.

The authors' method of solving the mathematical model is unnecessarily complex. Simplified methods of computing the average photosynthetic oxygenation rate, P, and the atmospheric oxygenation rate coefficient, K_2, are presented in this discussion. The method for P was previously available, having been used by the writer as early as 1950.[22] The simplified method for determining K_2 was recently developed and is presented here for the first time. These methods, because of their relative simplicity and accuracy, are superior to the computation method presented by the authors. Observations for only a 4-hour period will generally provide deficit data adequate for computation of P and K_2 by these techniques. Use of the simplified methods to determine P and K_2 directly and accurately should make it much easier to determine the three remaining unknowns–K_1, L_a, and R–with improved accuracy.

The authors' conclusion that Equation (2) is a satisfactory mathematical model for representing the DO budget in impounded waters is apparently based on the approximate fitting of curves to observed laboratory and simulated data. In this conclusion, they appear to have stated a *non sequitur*; it would not follow even from an exact fit of the computed curve to these data that the model is a satisfactory model for a prototype impoundment. The model may be valid for limited use, but the authors have not proved its validity. The model should be tested further under a variety of environmental conditions. If such testing supports the validity of the model for at least some waters and conditions, its general utility as a prediction tool will depend on the development of methods for predicting the five parameters, K_1, K_2, L_a, P, and R, for the range of environmental conditions expected.

In spite of many weaknesses in the authors' paper, their basic approach appears to be potentially useful and worthy of further research and development effort.

MANAGEMENT AND MEASUREMENT OF DO IN IMPOUNDMENTS

Authors' Closure

The writers wish to thank Thackston, Bella and Hull for their interesting comments. One of the reasons for original publication was to elicit the type of dialogue that, happily, developed.

EQUIPMENT

Hull's comments and criticisms of the equipment are useful, and the writers are sure that integration of their statements with Hull's will serve as a good guide to other investigators who wish to continue the development of experiments with this type of equipment. Hull questioned the light transmission of the clear plastic used in the light chamber. Data on the plastic used reported by Leifer[23] showed ''...that within the range of visible light at which algae function (400 mμ - 750 mμ), transmission is 90% or better.''

Several times in his discussion, Hull mentioned the advantages of collecting data over the shortest time-span. He apparently did not think this suggestion could be facilitated through the use of the equipment proposed by the writers. He stated, ''This type of improvement-one of procedure, not equipment-can be applied to bottles no less than to the authors' special containers.'' The writers do not disagree, but as noted in a previous paper,[9] the amplification circuit employed with this equipment does permit the detection of very small changes in DO concentration, thereby facilitating the obtaining of meaningful rates of DO concentration change, over the shortest possible time span. Also, by obtaining a continuous or semicontinuous record of DO concentration, the use of this equipment obviates the necessity of incubating several bottles, withdrawing one at a time for wet chemical analysis of DO concentration, to obtain a meaningful rate of DO concentration change.

METHOD OF OPERATION

Both Hull and Bella criticized the depth through which the data taken were said to apply, although Bella apparently overlooked the writers' statement that only the euphotic zone was involved. Even this statement was imprecise, however, and as Hull correctly pointed out, the data actually applied only to a single depth. The long-range objective for the use of this equipment, with modifications as necessary, was to take measurements over short time-spans, under relatively constant environmental conditions, at various times during a 24-hour cycle, and at various depths in a reservoir. These data would then be used to establish a relationship between environmental conditions and/or depth and the coefficients, (the writers agree with the objection of Hull and Bella over the use of the word ''constant''), in the mathematical model. The writers think that Hull caught the spirit of their intentions when he stated ''The values of the various 'constants' of Equation (2), determined by short-term observation using the three-probe technique, would be useful in an algebraic-summation model accounting for all oxygen sources and sinks.''

The writers fully realize that they have taken only the first few steps in the proverbial ''trip of a thousand miles,'' but because this ''trip'' was administratively cancelled, they thought investigators interested in this subject might find the results of their early travels useful. The writers hoped they had not overstated the case when they said, ''Future study will undoubtedly reveal further improvement in this approach, but this is a significant advance.''

MATHEMATICAL MODEL

In general, the criticisms of the proposed mathematical model fell into three categories. One, Thackston and Hull disagreed with our method of solution of the model; two, Bella disagreed with the form of the model; and three, all three discussers objected to our remarks concerning the physical meaning of the coefficients.

Method of Solution

The writers recognize that the best situation is one in which a perfect mathematical model can be written and solved using accurate, independent methods for evaluation of the coefficients. If, however, the coefficients are highly correlated, as Thackston noted is the case in the model under consideration, large errors may be introduced in the solution.

Thackston suggested a bottle method for the determination of L_a, and Hull offered other equations for determination of P and K_2. Both discussers then suggested that these values, once known, would facilitate the determination of the other coefficients in the model. Because of the high degree of correlation, however, the use of this approach would merely force compensating errors into the other three or four coefficients in the model. The writers, therefore, prefer establishing a best-fit criterion (non-linear least squares was used in the original paper), with no restraint on any of the five coefficients.

Hull was offended by the apparent complexity of the equations in the Appendix, but a computer program, which took only a few minutes of computer time, was easily prepared for this solution and when correct input data were used, gave good answers.

Form of Model

Bella suggested an entirely different form of the mathematical model. His suggestion of adopting a one-dimensional vertical approach is certainly worth considering. This would provide a more nearly perfect approximation to reality. Perhaps another approach might be to apply Equation (2) to a vertically distributed series of horizontal sections in which the DO could be assumed to be vertically mixed. The value of k_2 then would be interpreted as the net oxygen exchange across the imaginary boundaries of the sections.

Hull objected to Conclusion 1, which states that ''Equation (2) is a satisfactory mathematical model for representing the DO budget in impounded waters.'' The writers agree with Hull that this was too sweeping a statement. The word ''promising'' should be substituted for the word ''satisfactory'' in Conclusion 1.

Physical Meaning of Coefficients

All three discussers objected to the following paragraph: ''Through the years, research efforts have attempted to devise methods to evaluate and understand the physical meaning of the five constants (k_1, k_2, L_a, R and P) in Equation (2). While excessive concern over the precise meaning of these constants, has, with some justification, been criticized by Hull[2] and Ettinger,[3] this does not negate the utility of the overall model. The important feature of the model as used here is not the physical interpretation of the constants, but its ability to accurately reproduce what is occurring in nature.''

Although the writers accept Hull's comment that his remarks in Reference 2 were misinterpreted in the above paragraph, the writers think that all three discussers misunderstood the spirit of this paragraph. Hopefully, the following comments will clarify the writers' original position.

A mathematical model is an attempt to describe some natural phenomenon by means of a functional representation. Unless a function is available that exactly describes the natural process, the mathematical model will only be an approximation. Once a functional representation is assumed in a given situation, however, one is restricted, in terms of working with the model, to certain logical operations embodied in the discipline of mathematics. Therefore, once a mathematical model is assumed, one has to leave the physical realm and conform to the abstract realm of mathematical theory. The link between the formal mathematical operations and the natural phenomenon is formed by the definitions chosen for the coefficients and the interpretation of their magnitude.

One method of obtaining the magnitude of the coefficients is to establish a criterion such as a best-fit (non-linear least squares) criterion, which the writers have used. Although the writers knew Equation (2) was not an exact representation of the physical phenomenon, it seemed to provide a satisfactory approximation, and therefore was the functional representation chosen. The lack of physical exactness inherent in Equation (2) was the reason the writers were not originally overly concerned with the physical interpretation of the coefficients.

An example of this reasoning is as follows: Take,

$$Y = L_\alpha \, [1 - \exp(-k_1 t)] \qquad (42)$$

in which

Y = the organic biochemical oxygen demand at any time,

L_α = the total organic biochemical oxygen demand,

k_1 = the deoxygenation rate constant,

and t = time

as one possible mathematical model for the uptake of oxygen during the biological degradation of organic matter in a waste stream. For a waste that is a mixture of organic compounds, such as domestic sewage, this model is an approximation, because each organic compound in the mixed waste has its own total organic biochemical oxygen demand (L_α) and its own deoxygenation rate constant (k_1). These cannot be determined individually and summed, so some best-fit technique solves the model and evaluates the magnitude of the coefficients. The solution to this model is admittedly an approximation, but, because the magnitude of the coefficients are related to, even if not exactly representative of, the parameters of interest, their evaluation does allow comparisons and general conclusions. But what of the precise physical interpretation of the magnitude of the coefficients L_α and k_1? They do not even exist, except as a mathematical concept, so they can have no true physical meaning.

In the same way, in the writers' model, the coefficients k_1, L_α, R, P, and k_2 may or may not exist. Therefore, until mathematical functions representing the exact mechanism of deoxygenation, atmospheric reaeration, algal photosynthesis and algal respiration are known, the writers suggested that the chosen model was a useful mathematical tool. We did not think, however, that excessive concern over the precise physical meaning of these coefficients was necessary.

FURTHER THOUGHTS ON THE POTENTIAL PROBLEMS WITH THE MODEL

While the above remarks attempt to clarify the writers' original thinking, Thackston pointed out an important fact that had escaped the writers' attention. Thackston's Table 4, showing the high degree of correlation between the variables in this particular model, is very significant. This high degree of correlation means that the model under consideration is especially susceptible to trade-off between variables.

This susceptibility may be seen in the solution to the example problem in the Appendix. In the example, using the ''perfect'' (5-place) DO deficit data, perfect solutions were obtained; but using the slightly ''imperfect'' (rounded to 2 places) DO deficit data, a trade-off between the variables R and L_{α} occurred. This shows, as Thackston noted, that because of the high correlation between variables, the solution to this model is even more sensitive to slight changes in input DO deficit data than was stated in the original paper.

What does this mean in terms of the usefulness of Equation (2)? As stated above, the writers were willing to accept that the coefficients in the model had no precise physical meaning, but, if it were to be useful for prediction and comparisons, as was the stated objective in the original paper, the magnitude of these coefficients should at least reflect the physical situation.

We now wonder if this necessary relationship to the physical realm exists for this model. For example, would the solution to a series of DO deficit versus time curves established from various waters containing increasing concentrations of oxygen demanding organic matter yield increasing values for the coefficient L_{α}, or would trade-off occur? If trade-off did occur, and, for example, the solutions yielded randomly varying values for the coefficient L_{α}, doubt would be cast on the validity of the model. Only through further controlled experimentation, during which algal populations, bacterial populations, organic content, and wind velocity are varied over a wide range, and the effect on the magnitude of the coefficients in the model determined, can the seriousness of this potential problem be determined.

Hull was critical of the lack of extensive experimentation. We originally thought this lack was properly recognized by this statement: ''Finally, the DO probe equipment must be used in the field to gain experience with the influence of various natural environmental factors on the constants in the mathematical model and to determine whether good correlations [''relationships'' would have been a better word] between the model and data taken may be obtained under a variety of conditions.''

After continued reflection on the problem, however, the writers now think this statement was not strong enough. More controlled laboratory experimentation, as well as field work, is needed to determine whether this or any other model is sufficiently related to the physical realm to be useful. The writers are disappointed that they cannot perform the experiments that would supply the answers to the questions raised, particularly because the questions have such sweeping implications. Having determined to our own satisfaction that these doubts exist, however, we felt obliged to share our thinking with other investigators, with the hope that they might profit from it.

ALTERNATE APPROACH

If theoretical models cannot be developed, the authors suggest that the method of Churchill et al.[24] may be worthy of additional attention. This paper, which is specifically concerned with prediction of k_2, develops several predictive regression equations by correlating from historical data several parameters that intuitively appear important. The degree of correlation between pairs of variables in the various equations is then calculated, and the equation of choice is the one with the least correlation between variables. This consideration prevents the chosen equation from containing potential problems from trade-off. The technique has some drawbacks because the resulting prediction equation only applies to the situation for which it was developed. It has the major advantage, however, that the resulting equation is firmly related to the physical situation and contains independent variables.

SUMMARY

In conclusion, the writers would like to repeat their appreciation of the interest shown in their paper. All three of the discussers were kind enough to suggest that the paper had merit, at least within the limits of their reservations. The writers sincerely hope that this three-part dialogue, including some of the writers' later thinking, will prove useful to others studying in this area.

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PAPER NO. 5

INFLUENCE OF ENVIRONMENTAL FACTORS ON THE NITROGEN CYCLE IN WATER

J. DeMarco, J. Kurbiel, J. M. Symons,
and G.G. Robeck

This paper describes a controlled laboratory investigation that was performed to study the influence of various environmental factors on the nitrogen cycle in water. A nitrogen balance was consistently obtained by measuring only three of the nine possible valence states of nitrogen, using automated procedures for the nitrogen analyses. Ranges of initial seed, ammonia nitrogen, nitrite nitrogen, nitrate nitrogen, and organic concentration, and turbulence were then studied to evaluate their influence on the rate of ammonia nitrogen oxidation.

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INTRODUCTION

Understanding the behavior of impounded water is essential to the evaluation of any project involving streamflow regulation for water quality. Since 1962 the Engineering Activities Section of the Basic and Applied Sciences Program of the Federal Water Pollution Control Administration has been studying this subject, as specified in amended P.L. 87-88.[1] This study was one in a series, designed to investigate the influence of natural environmental factors on the nitrogen cycle.

The nitrogen cycle has been frequently discussed in the literature.[2-5] Figure 1 shows a simplified version of this cycle under aerobic conditions. There have been many studies of natural environments,[6-14] but few controlled studies to demonstrate the influence of environmental factors on the rate and extent of nitrification.

Environmental Influences

Environmental influences are important. For example, the oxygen demand of ammonia nitrogen is slightly over 4.5 milligrams O_2 per milligram NH_3-N. One study on the Grand River in Michigan[15] showed that the nitrification oxygen demand was three times the oxygen demand exerted by the discharge of organic matter. Therefore, oxidation of only 1 milligram of ammonia nitrogen per liter could stress the oxygen resources of a slowly reaerating reservoir.

The role that nitrogen plays in the synthesis of algae is also important. The literature discusses whether ammonia or nitrate nitrogen is more readily assimilated by algae for synthesis,[16,17] as well as their exact nitrogen requirements. Algal growth potential should be known for a given impoundment with a specific nitrogen content. Finally, the interference of ammonia nitrogen with chlorination must be considered if an impoundment is to be used for a water supply.

EQUIPMENT AND METHODS

The apparatus used to conduct the experiments has previously been reported by Symons and coworkers.[18] It consisted of several 5-gallon wide-mouth glass jars with provisions in each system for mechanical mixing through the top, a filtered air supply to maintain aerobic conditions, and a glass sampling siphon.

Synthetic river water (distilled water fortified with Yeast Nitrogen Base*) was used in all the experiments. The base supplied a variety of vitamins, minerals, amino acids, and ammonium sulfate, which served as the source of ammonia nitrogen.

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*Difco Yeast Nitrogen Base, a product of Difco Labs., Inc., Detroit.

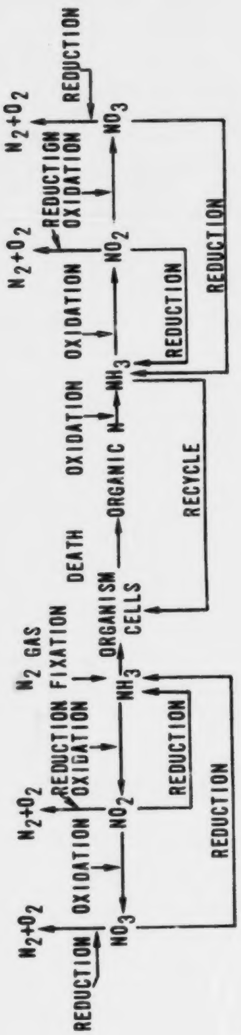
REDUCTION
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Figure 1. Summary of nitrogen cycle, after Hutchinson.

The system was buffered with sodium bicarbonate and monopotassium dihydrogen phosphate. The final pH was 7.5 and the alkalinity was approximately 200 milligrams per liter as $CaCO_3$. All the tests were conducted at 23° to 25°C. The pH and temperature were in the range suitable for growth of the nitrifying populations.

The seed used in the system was obtained from an operating bench-scale activated sludge plant that was actively nitrifying. The mixed liquor taken from the activated sludge plant was washed twice with distilled water to remove soluble carryover products, and then placed in a blender* to disperse the floc. The volatile suspended solids concentration, determined by a membrane filter technique,[19] was used as a basis for controlling the initial solids concentration desired. The relatively low volatile suspended solids (VSS) used in the experiments approximated the field conditions of a stream or impoundment.

Samples were periodically taken for pH, alkalinity, COD, dissolved oxygen, four nitrogen forms (ammonia, nitrite, nitrate, and organic nitrogen), and both ammonia and nitrite nitrogen oxidizing microorganism populations.

The analytic techniques used to determine the nitrogen concentrations and the nitrifying populations were somewhat unique. They have been previously reported.[18,20] The nitrogen concentrations were either wholly or in part determined by use of an automatic analyzer.** The nitrifying populations were determined using a modified MPN procedure that was an adaptation and extension of the method presented by Lewis and Pramer.[21] The analytic methods used for pH, alkalinity, and COD are described in *Standard Methods*.[22]

RESULTS

Typical Nitrification Patterns

In all the investigations, the four nitrogen forms were closely monitored to ascertain whether or not a nitrogen balance could be obtained. Figure 2 shows the typical nitrification patterns obtained. The excellent recovery of all four nitrogen forms is indicated by the nearly horizontal ΣN curve. The ΣN is the arithmetic summation of all the forms for each day. It shows that all the nitrogen was

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*Waring blender, a product of Waring Products Co., New York.

**AutoAnalyzer, a product of Technicon Corporation, Tarrytown, New York.

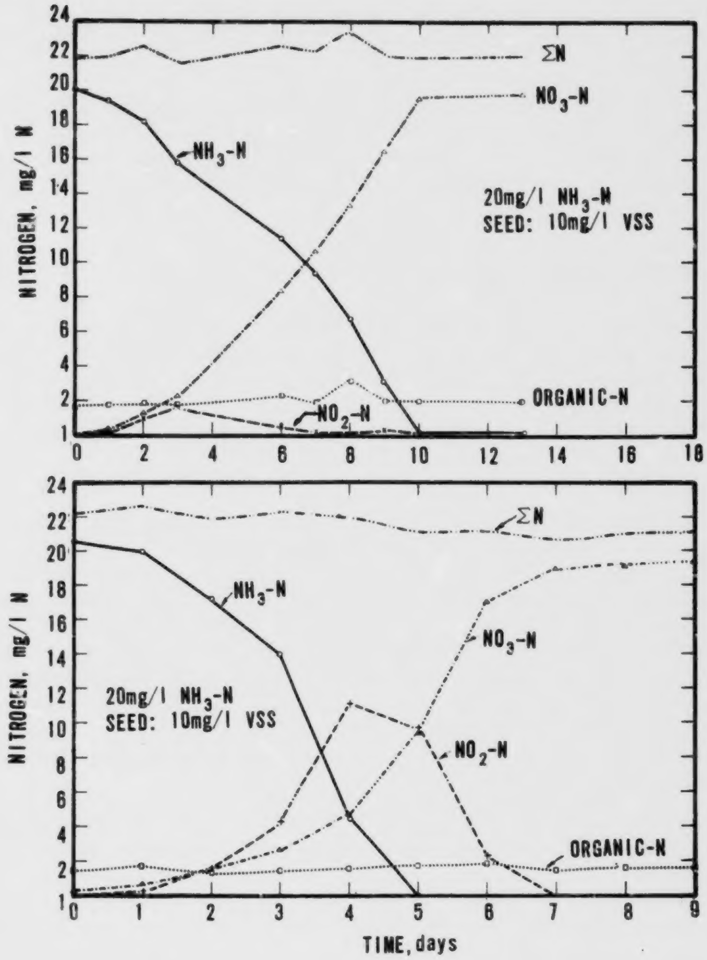

Figure 2. Typical nitrification patterns.

continuously accounted for, although only three of the nine possible valence states for nitrogen were monitored. The results of 28 of 32 runs produced ΣN curves similar to those shown in Figure 2. The organic nitrogen remained nearly level in all the experiments, except when organics were added to the test system. Figure 2 shows the classical nitrification pattern, with the oxidation of ammonia nitrogen, and subsequent production and disappearance of nitrite nitrogen concurrent with the production of nitrate nitrogen.

After the test equipment and analytic procedures were successfully demonstrated, several environmental factors were studied. They included: initial seed and NH_3-N concentrations, turbulence, initial NO_2-N, NO_3-N, and organic concentrations.

Seed Concentration

The presence of an adequate nitrifying population is one of the most important factors in nitrogen transformations. The numbers of actively nitrifying bacteria present when ammonia nitrogen is first introduced to a system should affect the rate of nitrification because nitrifying bacteria have long generation times.[8]

Seed concentrations were set up by varying the initial volatile suspended solids concentration from 5 to 25 milligrams per liter in five different systems. Initial counts of the ammonia and nitrite oxidizing populations verified that an increase in the volatile suspended solids concentration effected the same order of increase in the actual seed concentration. The nitrifying organism counts showed that when the volatile suspended solids was increased from 5 to 25 milligrams per liter, an approximate five-fold increase also occurred in both the ammonia and nitrite oxidizing populations.

Figure 3 shows that as the initial seed concentration increased, the initial rate of ammonia nitrogen oxidation increased. Also, the time for complete oxidation of an initial concentration of 10 milligrams of ammonia nitrogen per liter was decreased from 7 days in the system with a lower seed concentration (5 milligrams per liter as VSS) to 4 days in the system with a higher seed concentration (25 milligrams per liter as VSS).

Ammonia Nitrogen Concentration

A suitable substrate, in adequate concentration, is required for maintaining and increasing a nitrifying population. Inorganic nutrients are essential to the metabolic pattern of autotrophs. Nitrifiers specifically require inorganic nitrogen as an energy source to permit growth.

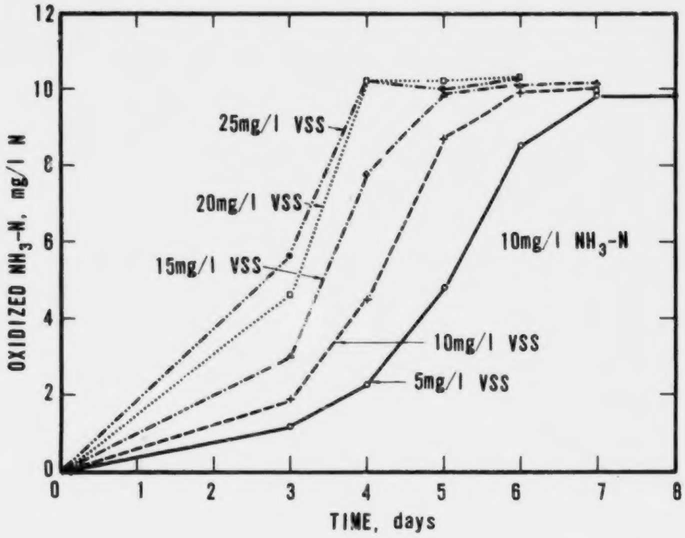

Figure 3. Influence of seed variation on nitrification.

In utilizing ammonia nitrogen as a substrate for growth, an oxygen demand exists that may create a stress on the oxygen budget of a natural environment. The presence of a high concentration of ammonia nitrogen ultimately causes a greater oxygen demand than a low amount, but the rate of demand depends on the rate at which the organisms utilize ammonia nitrogen. Therefore, various initial concentrations of ammonia nitrogen were investigated to determine their effect on nitrification.

Figure 4 shows the effect of varying initial ammonia nitrogen concentrations on nitrification with the initial seed concentration constant (10 milligrams per liter VSS). The initial ammonia nitrogen concentrations (5 to 20 milligrams per liter NH_3-N) did not affect the rate of nitrification. For instance, about 2 milligrams of ammonia nitrogen per liter had been oxidized by the second day of the test in all four systems. Therefore, in a stream or impoundment the rate of oxygen demand for the oxidation of ammonia nitrogen would be the same over this range of ammonia nitrogen concentration. The ultimate oxygen demand of the 20 milligrams per liter ammonia nitrogen system would be four times greater than the 5 milligrams per liter ammonia nitrogen system. A flowing

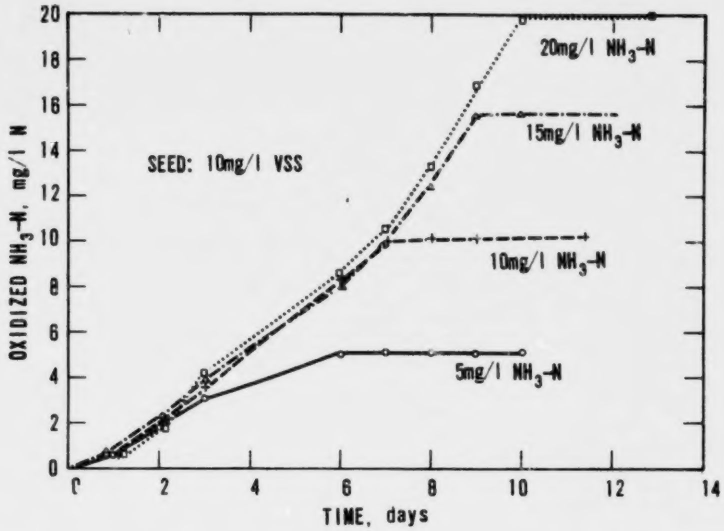

Figure 4. Influence of ammonia variation on nitrification.

stream with adequate reaeration could possibly assimilate high or low ammonia concentrations, however, at the same rate of oxygen consumption. The high ammonia nitrogen condition would simply cause an oxygen demand for a longer reach of the stream.

This finding, that ammonia nitrogen in the range of 5 to 20 milligrams per liter does not influence the rate of nitrification, is in agreement with results at the Water Pollution Research Laboratory at Stevenage.[8] This work showed that ammonia nitrogen concentrations above 2 milligrams per liter (saturation constant = 0.18 milligram per liter at 21°C) did not affect the respiration rate of a nitrifying sludge.

Relative Turbulence

If turbulence has a positive effect on ammonia nitrogen oxidation, a flowing stream would be expected to nitrify faster than a quiescent system such as an impoundment. Figure 5 represents data from two experiments showing that ammonia is oxidized at a faster rate under high turbulence. The upper graph shows that with high turbulence, 4 days less were required to oxidize completely 20 milligrams of ammonia nitrogen per liter. The lower graph

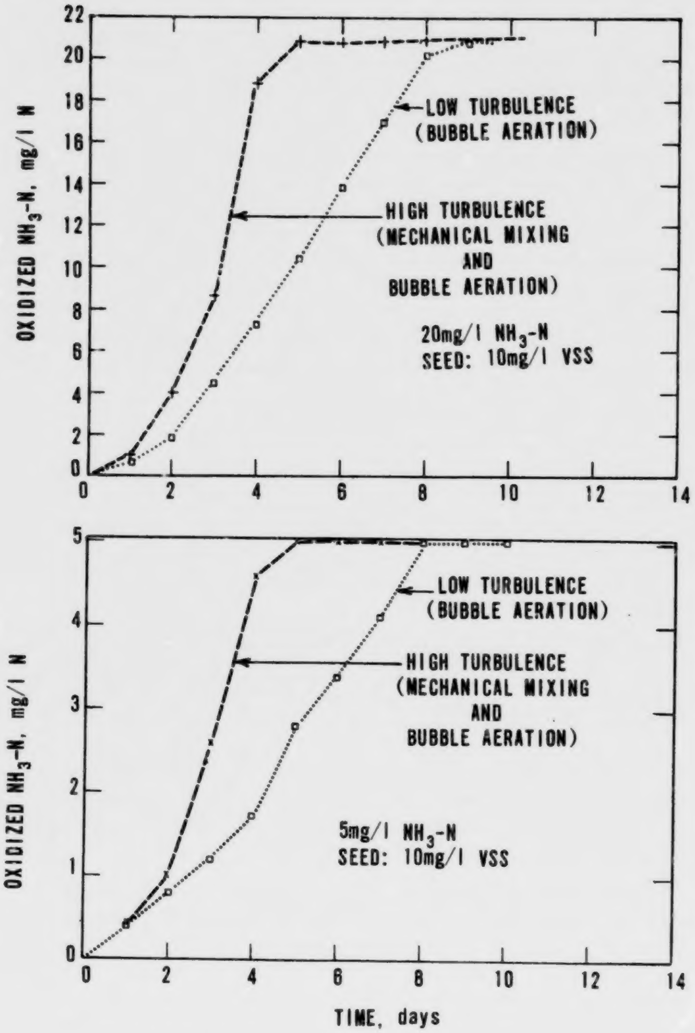

Figure 5. Influence of turbulence on nitrification.

shows that at an initial ammonia nitrogen concentration of 5 milligrams per liter, 3 days less time was required for the high-turbulence system to completely oxidize the ammonia nitrogen. In each test reaction times were compared to those in a low-turbulence system.

The terminology is not quantitative. A system with mechanical mixing plus bubble aeration is considered to have high turbulence. Low turbulence refers to a system with bubble aeration only. Bubble aeration is necessary to maintain aerobic conditions.

The results are logical. The high-turbulence system should prevent microorganisms from settling, thereby keeping them in more intimate contact with the nitrogenous substrate in solution. Although initial ammonia nitrogen concentration has been shown not to affect the rate of nitrification, the oxidation rates in the upper graph of Figure 5 cannot be compared to those in the lower graph of Figure 5 because the seed for each experiment was obtained from the activated sludge pilot plant at different times.

Interference of Other Substances

As a natural environment includes substances that might inhibit the growth of nitrifiers, the effect of interfering substances such as nitrite, nitrate, and glucose was investigated.

NITRITES AND NITRATES

High nitrite concentrations, 200 to 2,500 milligrams per liter, have been reported to inhibit ammonia nitrogen oxidation.[23,24] This phase of the study was designed to determine whether low nitrite nitrogen concentrations would cause inhibition. The lower graph of Figure 6 shows the results of four systems. All the systems initially received 10 milligrams volatile suspended solids per liter and 20 milligrams ammonia nitrogen per liter, and nitrites and nitrates as noted.

The nitrate nitrogen had no effect on the rate of ammonia nitrogen oxidation. Therefore, the system that contained nitrates and nitrites became a study of the effect of the initial presence of 5 milligrams of nitrite nitrogen per liter on nitrification. Compared to the control, the system with 5 milligrams nitrite nitrogen per liter had a 2-day lag period. The system with 10 milligrams initial nitrite nitrogen per liter had a more extended lag period.

The upper graph of Figure 6 shows the nitrogen oxidation pattern for the control run. The control did not receive any initial nitrite nitrogen, but it did accumulate in the system from the oxidation of ammonia nitrogen. The upper

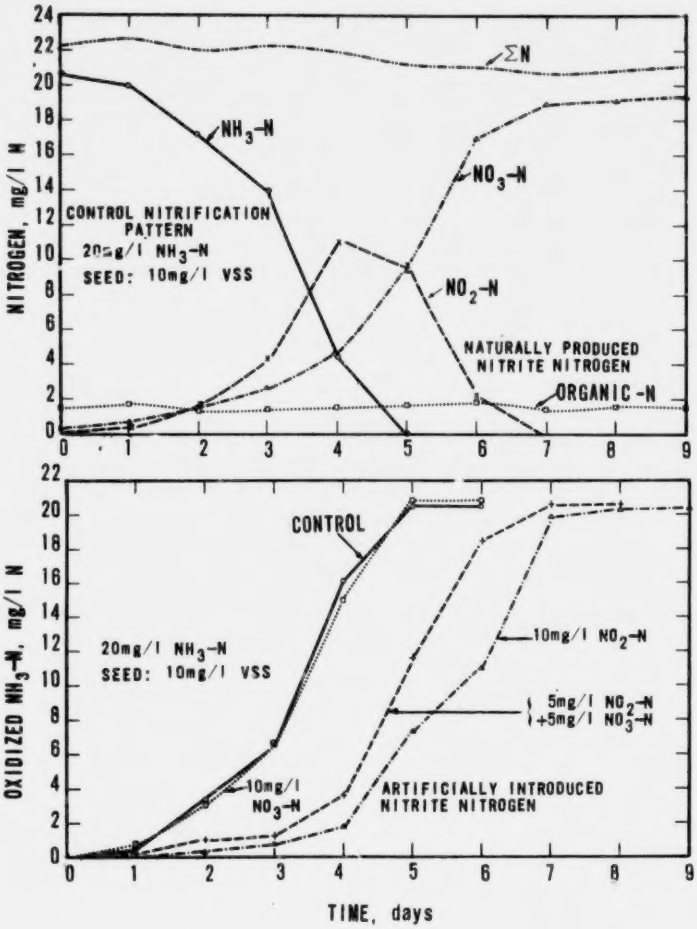

Figure 6. Influence of nitrites and nitrates on nitrification.

graph shows that from the third to the fifth day the nitrite nitrogen concentration was above 5 milligrams per liter. It was 11 milligrams per liter on the fourth day. There was no lag effect over this period. This is shown by the smooth progression of ammonia nitrogen oxidation. These results indicated that the artificial introduction of nitrites at the beginning of a test caused a lag effect, but a system that naturally produced nitrite nitrogen did not.

The nitrifying population counts were examined and the ammonia oxidizing populations were found to be above 140 organisms per 100 milliliters before active oxidation of ammonia nitrogen was begun in the two systems with initial nitrites. The system that naturally produced nitrite nitrogen usually had ammonia oxidizer counts far above this level before appreciable nitrite nitrogen accumulated. Thus, the inhibitory effect caused by nitrites, in the systems studied, may have been related to the number of active ammonia oxidizers present at the time that nitrites were introduced. This was not proved conclusively, however.

ORGANICS

The general concept of the two-stage BOD reaction that appears throughout the literature depicts nitrification occurring after the oxidation of carbonaceous matter. Various workers have indicated that nitrification and carbonaceous oxidation may occur simultaneously.[25,26] In this study glucose was added to two systems to determine the effect of an easily oxidizable organic substance on the oxidation of ammonia nitrogen.

The graph of COD versus time in Figure 7 shows the duration glucose was present. No attempt was made to maintain the glucose level at the initial concentration; it was a batch experiment. The graph of total COD versus time in Figure 7 shows that the control run had a background COD of 16 milligrams per liter. The initial COD was higher in the other systems because of the glucose present. Both glucose systems showed a rapid decline in COD over the first 2 days because of biologic assimilation of glucose. After the third day the total COD probably resulted from the heterotrophic organisms produced and the background COD. This accounts for the much slower decrease in total COD afterwards.

The graph of oxidized ammonia nitrogen versus time in in Figure 7 shows the results of systems with no glucose, 100 milligrams glucose per liter, and 200 milligrams glucose per liter initially present. The 100 milligrams glucose per liter curve compared well, at first, with the no-glucose system, but after the second day the rate of ammonia nitrogen oxidation decreased relative to the no-glucose system. The oxidized ammonia nitrogen in the 200 milligrams glucose per liter system had a 2-day lag.

In Figure 8 the lower graph shows the nitrogen pattern for the 100 milligrams glucose per liter system. The

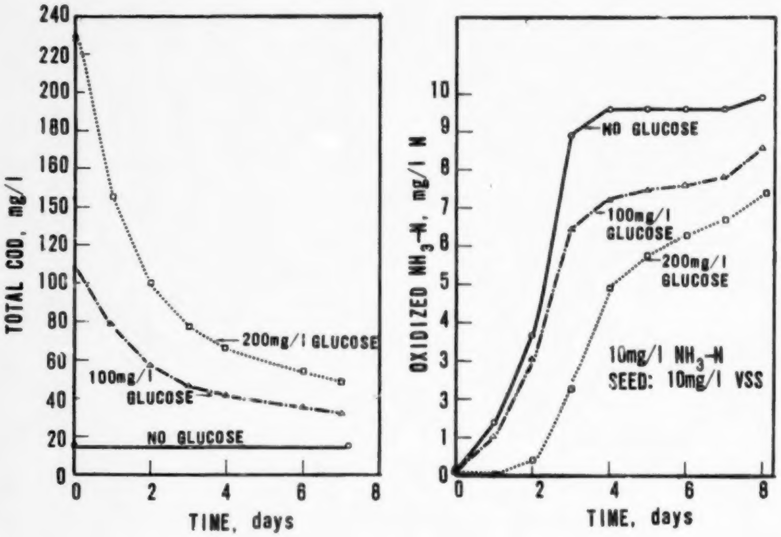

Figure 7. Influence of organics on nitrification.

glucose added to the system caused a rapid proliferation of the heterotrophic population; the total count rose from 5.6×10^5 to 8.8×10^9 organisms per milliliter during the first day. These organisms utilized ammonia as a source of nitrogen for cell synthesis. This is reflected in the lower graph of Figure 8 by a rapid increase of 5.9 milligrams per liter in the organic nitrogen curve. This accounted for most of the decrease in ammonia nitrogen during this period. The increase in the sum of the nitrite nitrogen plus nitrate nitrogen on the first day showed that, though most of the ammonia nitrogen was converted to organic nitrogen, there was oxidation of ammonia nitrogen by the autotrophs. This indicated that simultaneous oxidation of nitrogenous and carbonaceous matter occurred. Also, the amount of ammonia nitrogen oxidized in the 100 milligrams glucose per liter system compared well with that in the no-glucose system (see Figure 7). Therefore, the glucose had no effect on the amount of ammonia nitrogen oxidized during this period.

After the second day the lysed cells were the only appreciable source of ammonia nitrogen. Beyond this time the rate of ammonia nitrogen oxidation of the 100 milligrams glucose per liter system decreased (see Figure 7)

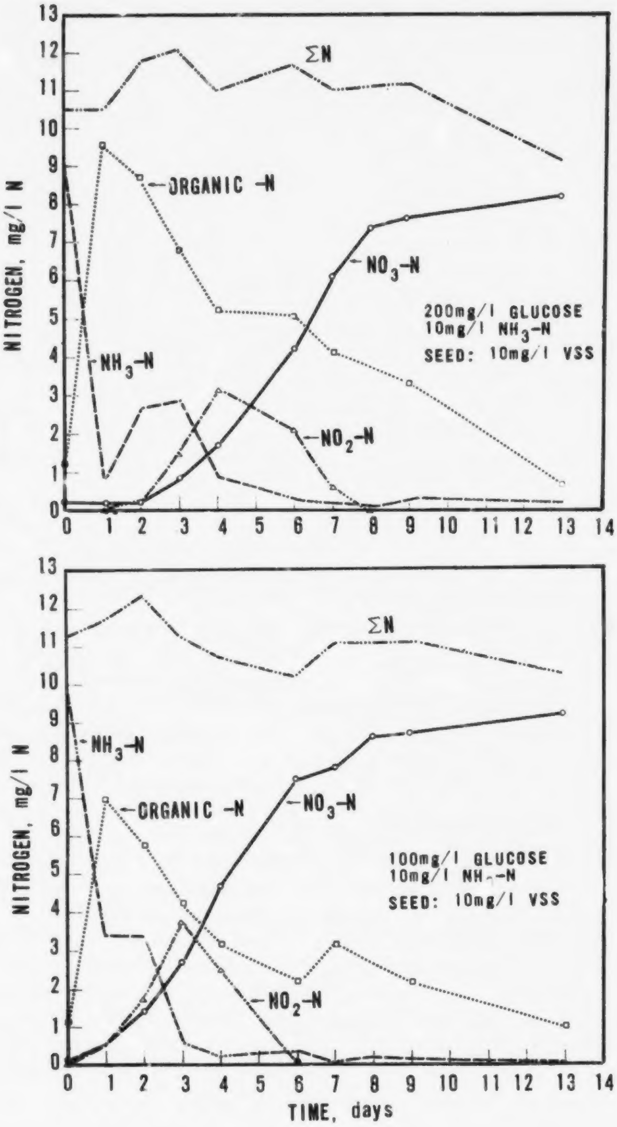

Figure 8. Influence of organics on nitrification.

because its availability was limited by the death and subsequent lysis of the heterotrophs. Although the glucose was only present during the first 2 days of the test and did not initially affect the oxidation of ammonia nitrogen, the conditions produced by the glucose decreased the rate of oxidation in the latter days of the 100 milligrams glucose per liter system.

The upper graph of Figure 8 shows the nitrogen oxidation for the 200 milligrams glucose per liter system. This increased glucose concentration caused additional depletion of the ammonia nitrogen by conversion to organic nitrogen. In this instance 8.4 milligrams ammonia nitrogen per liter was converted to organic nitrogen, leaving a substrate-limiting condition for the autotrophs after the first day of the test. The absence of any oxidized ammonia nitrogen (nitrite plus nitrate nitrogen) during the first 2 days verified the existence of a substrate-deficient condition. This explained the 2-day lag. When the ammonia nitrogen concentration increased to 2.7 milligrams per liter because of the heterotrophic degradation, the autotrophs began oxidation to nitrite and nitrate nitrogen. After the fourth day the rate of cell lysis controlled the rate of ammonia nitrogen oxidation because there was no excess ammonia nitrogen in solution.

Bacterial Enumeration

The evaluation of the nitrification potential of a natural environment is desirable. A correlation between the number of nitrifiers present and the expected degree of nitrification seemed a possible method of prediction. Bacterial counts were correlated with various nitrification patterns.

Bacterial correlations with chemical changes during nitrification have been investigated and reported.[18] Figure 9 shows a typical example of the excellent correlations between nitrifying populations and the chemical transformations of nitrogen. In 16 of the 18 systems the resulting correlations were similar to those shown in Figure 9.

Although the bacterial and chemical correlations were excellent in showing trends, a theoretic thermodynamic calculation of cell yield, as reported by McCarty,[27] indicated a low efficiency in counting the nitrifiers. For example, one calculation indicates that in oxidizing 10 milligrams of ammonia nitrogen per liter approximately

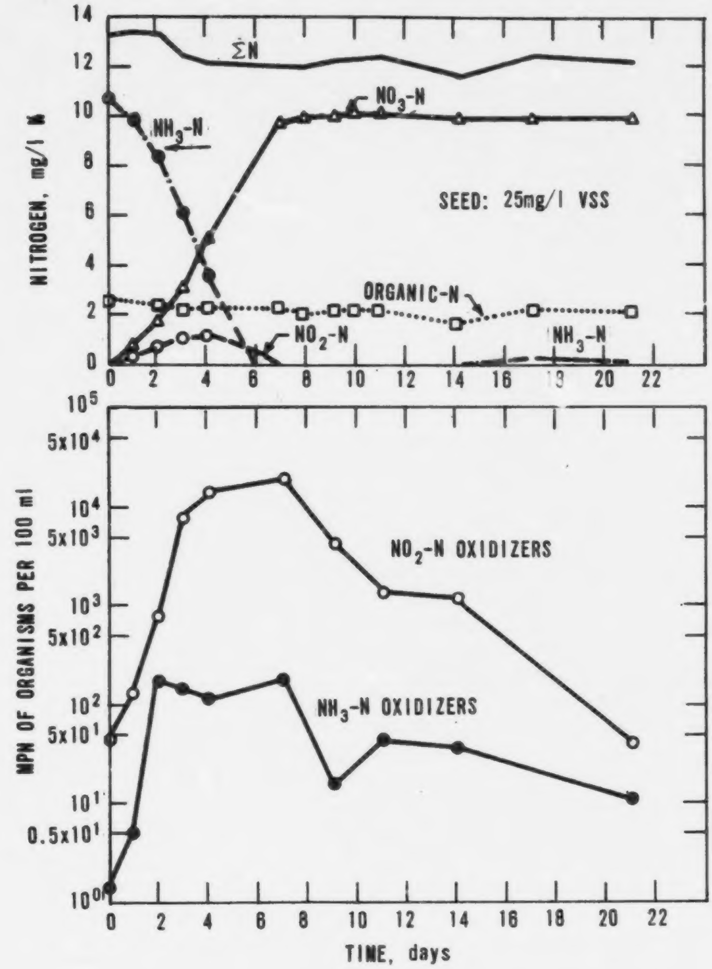

Figure 9. Correlation of bacterial and chemical data during nitrification.

0.3 milligram of cells by dry weight per 100 milliliters are produced. Oxidation of 10 milligrams of nitrite nitrogen per liter yields approximately 0.1 milligram of cells by dry weight per 100 milliliters. Using the dry weight of a cell as 1×10^{-10} milligram, approximately 1×10^{9} cells per 100 milliliters of both ammonia and nitrite oxidizers would be created during oxidation. As the actual counts of both groups varied between 1 x 10^2 and 1 x 10^4 organisms per 100 milliliters, a low efficiency of counting was indicated.

The combination of low efficiency and the time required to obtain the counts (4 weeks for the ammonia oxidizers and 6 weeks for the nitrite oxidizers) eliminates this modified MPN technique. If this approach to evaluating the nitrification potential is to be fruitful, an improved counting technique must be developed.

CONCLUSIONS

1. Nitrogen balance was attained in a clean, aerobic system by measuring only three of nine possible valence states of nitrogen, using an automated procedure for nitrogen analyses.

2. An increase in initial nitrifying seed concentration caused an increase in the initial rate of ammonia oxidation.

3. An initial concentration of ammonia nitrogen between 5 and 20 milligrams per liter had no effect on the initial rate of oxidation of ammonia nitrogen.

4. High turbulence as defined in this study caused an increase in the initial rate of ammonia oxidation compared to low turbulence.

5. Presence of 5 and 10 milligrams of nitrite nitrogen per liter caused the oxidation of ammonia nitrogen to slow when introduced at the beginning of the tests, but not when naturally produced from the oxidation of ammonia nitrogen.

6. Nitrate nitrogen had no effect on the initial rate of ammonia nitrogen oxidation.

7. Glucose had several effects on the initial rate of ammonia nitrogen oxidation:

a. If the initial ammonia nitrogen was not completely assimilated by the heterotrophic population during organic oxidation, simultaneous oxidation of carbonaceous and nitrogenous matter took place without influence on the initial rate of nitrification.

b. If the ammonia nitrogen was completely assimilated during organic oxidation, nitrification could not begin until some ammonia nitrogen was released back into solution by the death and lysis of heterotrophic bacteria.

c. In either a or b, when the ammonia nitrogen was eliminated from solution and the only source of ammonia nitrogen for nitrification was from cellular organic nitrogen, the rate of nitrification was controlled by the cell death rate. This was usually slow.

8. Bacterial counts were correlated with chemical changes to obtain a method of evaluating the nitrification potential of a given environment.

9. Attempts to develop a predictive tool for evaluating the nitrification potential were unsuccessful because of the low efficiency of counting and length of time required for enumerating the nitrifying populations.

ACKNOWLEDGMENTS

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PAPER NO. 6

BEHAVIOR OF SYNTHETIC ORGANICS IN STRATIFIED IMPOUNDMENTS

J. DeMarco, J.M. Symons, and G.G. Robeck

This paper discusses a laboratory investigation performed with an apparatus designed to thermally and chemically stratify 500 gallons of test media. Little Miami River water served as the natural test water, to which known amounts of selected synthetic organics were added. The behavior of the synthetic detergents sodium lauryl sulfate and linear alkylate sulfonate and of the pesticide 2,4-D were investigated under various environmental conditions that exist in stratified impoundments.

The correct reference citation is:

DeMarco, J., J. M. Symons, and G. G. Robeck. "Behavior of Synthetic Organics in Stratified Impoundments." Journal American Water Works Association, 59, No. 8, 965-976. August 1967.

INTRODUCTION

Recent interest in the use of low-flow augmentation for water quality control has increased the need to understand the behavior of impoundments. If impounded water is to upgrade stream conditions at selected intervals, then the quality of the released water must be higher than that of the receiving stream. Therefore, the determination of whether various contaminants are degraded and disappear, or persist and concentrate is required. To supply this information on some of the commercially important synthetic organics in the United States, a study was made of

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the behavior of synthetic organics in simulated stratified impoundments.

Occurrence of Synthetic Organics

Synthetic organics are a rapidly expanding, widely used group of substances that enter surface waters in a variety of ways, including land runoff,[1] accidental spills,[2] and air transport.[3]

The soap and detergent manufacturers are aware of the effects of synthetic organics on water quality. Recently, they developed a synthetic detergent that was less resistant to biodegradation. The result was industry-wide conversion from a partially biologically undegradable alkyl benzene sulfonate (ABS) to the more easily degradable linear alkylate sulfonate (LAS) in June 1965. Interest in the biodegradability of LAS was demonstrated by the investigations that have been made in activated sludge systems,[4,5] septic tanks and oxidation ponds,[6] and the laboratory.[7]

Pesticides are another example of synthetic organics that have become increasingly important. Various workers have documented the appearance of a wide variety of pesticides in water,[8-12] and their effect on water resources.[13,14] For example, Faust and Aly cite incidences of pollution caused by organic pesticides,[2] and the recent death of large numbers of fish in the lower Mississippi River was attributed to endrin.[15]

Persistence of Synthetic Organics

Although the effect of impoundments on the assimilative capacity of organics has been studied,[16] the persistence of synthetic organics under conditions that frequently occur during stratification has not been widely investigated. These conditions often involve warm, aerobic upper layers of water and cold bottom layers that are either very low in dissolved oxygen, anaerobic, or contain a reducing environment. These environments were produced in the laboratory for this investigation.

EQUIPMENT AND METHODS

Laboratory Apparatus

The laboratory apparatus[17] was designed to simulate thermal and chemical stratification in a typical stratified impoundment. The apparatus (Figure 1) consists of an aluminum tank, 3 feet in diameter and approximately 12.5 feet high, which holds 500 gallons (2,000 liters) of test

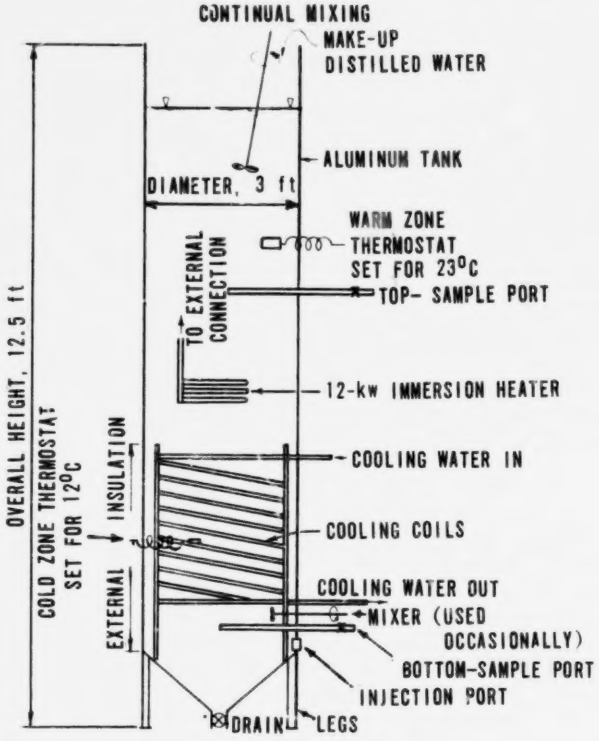
CONTINUAL MIXING
MAKE-UP DISTILLED WATER
ALUMINUM TANK
DIAMETER, 3 ft
WARM ZONE THERMOSTAT SET FOR 23°C
TOP- SAMPLE PORT
TO EXTERNAL CONNECTION
12-kw IMMERSION HEATER
OVERALL HEIGHT, 12.5 ft
COLD ZONE THERMOSTAT SET FOR 12°C
EXTERNAL INSULATION
COOLING WATER IN
COOLING COILS
COOLING WATER OUT
MIXER (USED OCCASIONALLY)
BOTTOM-SAMPLE PORT
INJECTION PORT
DRAIN
LEGS

Figure 1. Simulated stratified impoundment.

solution. The water depth is approximately 10 feet, with sample ports 3 and 9 feet below the surface. The upper two-thirds of the water column is heated by a thermostatically controlled 12-kw immersion heater, and the water is circulated continuously by a small laboratory mixer. The lower one-third of the water column is cooled by a thermostatically controlled refrigeration unit. An injection port and mixer are located at the bottom of the tank to permit the introduction and dispersion of substances into the bottom water zone. Four such simulated stratified impoundments were available so that concurrent tests could be performed.

Operational Procedure

At the beginning of a test, each tank being used was filled with Little Miami River water (see Table 1) and

Table 1. CHARACTERISTICS OF FIVE TEST WATERS

Characteristic	Range (In mg/l, except as otherwise indicated)
pH	7.5 - 8.5
Alkalinity as $CaCO_3$	124 - 206
Total hardness as $CaCO_3$	180 - 270
Turbidity–Jackson units	4 - 27
Total COD	10 - 20
NH_3-N	<0.2 - 0.25
NO_2-N	Trace - 0.02
NO_3-N	0.6 - 3.2
Organic-N	0.4 - 0.6
Total suspended solids	7.5 - 37
ABS	0.2 - 0.3
Bacterial count, organisms/ml	5.6×10^5 - 2.3×10^6
Total organic chlorine, μg/l	0.2

allowed to stand for 24 hours to permit sedimentation of the larger suspended solids. The sediment was then flushed from the bottom of the tank. A low suspended solids level was desired to minimize any possible loss of the synthetic organic tested by adsorption on particulate matter. A specified amount of synthetic organic was added to the water and dispersed throughout with a high-speed mixer. The water column was then thermally stratified. When a cold-deoxygenated environment was under investigation, glucose was added to the bottom water to promote more rapid deoxygenation of this zone.

Synthetic organic concentration, dissolved oxygen, and temperature were monitored at frequent intervals throughout each test. Total bacterial counts, total suspended solids (TSS), COD, and the nitrogen series were measured occasionally during the study. Make-up distilled water was added throughout each test to account for evaporation losses. Prior to each test, the tanks were treated with a chlorine solution of 5 milligrams per liter to remove any sidewall growths that may have accumulated. After a 4-hour contact time the tanks were thoroughly rinsed. This precaution was taken to avoid the possible influence of a previous test.

Use of Control

A control was used for the pesticide studies because of the possibility that some mechanism other than biodegradation, such as adsorption, volatilization, photochemical and thermal decomposition, and chemical hydrolysis, might cause significant losses during a long-term study. The test media for the control was deionized water in which a chlorine residual of 0.5 milligram per liter was maintained at all times.

If the synthetic organic losses in the control were of the same magnitude as those of the river water system, biologic degradation could not be responsible for pesticide removal. A large difference in the magnitude of losses, on the other hand, would be indicative of biologic degradation or adsorption on particulates in the river water. Adsorption as a mechanism of removal was evaluated by the material balance procedure described later. Biologic degradation or any of the other possibilities could be detected only by differences.

Analyzing Detergents and Pesticides

The analyses for synthetic detergents were performed by using the methylene blue procedure[18] that was automated for use on an analyzer.*

For pesticides, samples were analyzed for 2,4-D content with a gas chromatograph equipped with a microcoulometric detector. The samples were collected in glass containers to minimize contamination and adsorption. Analyses were either started immediately or the samples were preserved by acidification and storage at refrigerator temperature, about 4°C.

The analysis for 2,4-D was adapted from a method proposed by Gentzendaner[19] who described a procedure for the direct chromatographic analysis of the herbicide Dalapon without prior esterification. Samples were acidified with phosphoric acid, saturated with sodium chloride, and then extracted with two successive portions of diethyl ether. The concentrated solvent extract was then injected directly into a chromatographic column packed with diethylene glycol adipate. Nearly quantitative extraction of the 2,4-D from the water was obtained, although recovery from the chromatographic operation amounted to about 53 percent. Corrections for the low recoveries through the column were made by concurrently analyzing reference samples and making the appropriate corrections. The procedure was capable of determining a concentration of 0.05 microgram per liter. The standard deviation for 2,4-D based on the recovery of reference extracts at 50 micrograms per liter was ± 17 percent.

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*AutoAnalyzer, a product of Technicon Corporation, Tarrytown, New York.

Balance Procedure for Detergents

Upon completing each synthetic detergent test, a check was made of the possibility that any disappearance might be attributed to adsorption. The liquid test media was siphoned from the tank to avoid disturbing the sediment. The deposited sediment was collected, dried, weighed, and analyzed for total milligrams of the respective synthetic detergent. Calculation of the difference between initial and final total suspended solids concentrations of the test media gave an approximate total amount of sediment that could be recovered from the tanks. This calculated amount of sediment recovered was used as a guide to verify that most of the sediment was collected at the end of each run.

Balance Procedure for Pesticides

After each pesticide run a procedure was followed to account for any pesticide lost because of adsorption. The water was siphoned from the simulated impoundment to a level just above the bottom of the tank, with care being taken not to disturb the deposited sediment. Samples of the water were taken at various depths to determine the total weight of pesticide remaining in the liquid phase at the end of the test. The sediment was collected in a glass jug and analyzed for pesticide content. A calculation of the theoretic amount of sediment deposited was used to verify that most of the solids were collected. The interior sidewalls and appurtenances of the simulated impoundment apparatus were pressure-rinsed with 1 gallon of acetone to dry the tank, and then 1 gallon of hexane to extract any adsorbed pesticide. The solvents were collected and analyzed for pesticide content. A summation was made of the total micrograms of each pesticide found in the deposited sediment and solvent.

RESULTS

The synthetic organics investigated were selected because they are commonly used in the United States and have been reported to appear in surface waters. Table 2 lists the synthetic organics concentrations and types of stratified zones used. The top waters were warm and aerobic; the bottom waters were either cold and aerobic or cold and deoxygenated.

Table 2. CONCENTRATION OF SYNTHETIC ORGANICS AND TYPES OF ZONES

Synthetic organic	Approximate initial concentration	Types of zones studied
Detergents		
Sodium lauryl sulfate	5 mg/l, 3 mg/l	Warm-aerobic Cold-aerobic Cold-deoxygenated
Linear alkylate sulfonate	5 mg/l	Warm-aerobic Cold-aerobic Cold-deoxygenated
Pesticides		
2,4-dichlorophenoxyacetic acid	50 µg/l	Warm-aerobic Cold-aerobic Cold-deoxygenated

The modified simulated stratified impoundments used in this study promoted the same extent of thermal and chemical stratification as previously reported for the original units.[17] Figure 2 is a typical temperature profile for any of the modified test units. The warm-aerobic zones had an average temperature of 22° to 26°C and an oxygen content greater than 2 milligrams per liter. The cold-aerobic zones had the same oxygen content range, but the average temperature varied between 10° and 13°C. The cold-anaerobic zones had a range of 10° to 13°C, but an oxygen content below 0.3 milligram per liter at all times and usually below 0.1 milligram per liter.

Sulfides and ferrous iron were measured in the aerobic and deoxygenated environments in an attempt to show the presence of reducing bottom-water conditions. No reduced iron or sulfur was found. Comparison of the nitrate-nitrogen data of the upper and lower waters showed that the aerobic upper waters maintained the initial nitrate-nitrogen concentration throughout the test. The deoxygenated bottom waters reduced the nitrate-nitrogen concentration, however, to an average of 0.1 milligram per liter upon deaeration and maintained this level throughout the test. Although the bottom waters could not be considered strongly reducing, the environmental conditions reduced and maintained the nitrates throughout the test period.

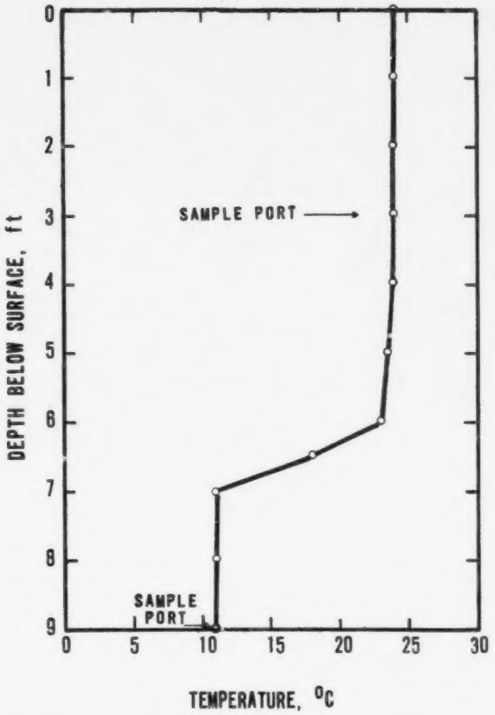

Figure 2. Typical temperature profile - simulated stratified impoundment.

Sodium Lauryl Sulfate (SLS) Removal

The lower graph of Figure 3 shows that the SLS was 96 percent degraded in approximately 18 hours under warm-aerobic conditions, whereas 92 percent degradation took approximately 5 days under cold-deoxygenated conditions.

Because both low temperature and low dissolved oxygen concentration contribute to decreasing metabolic activity, to cause a slower rate of degradation, the test was repeated. In this case, bottom waters were not deoxygenated so that the behavior in the warm-aerobic zone could be compared to a cold-aerobic zone and the relative effect of temperature and dissolved oxygen evaluated. The two curves

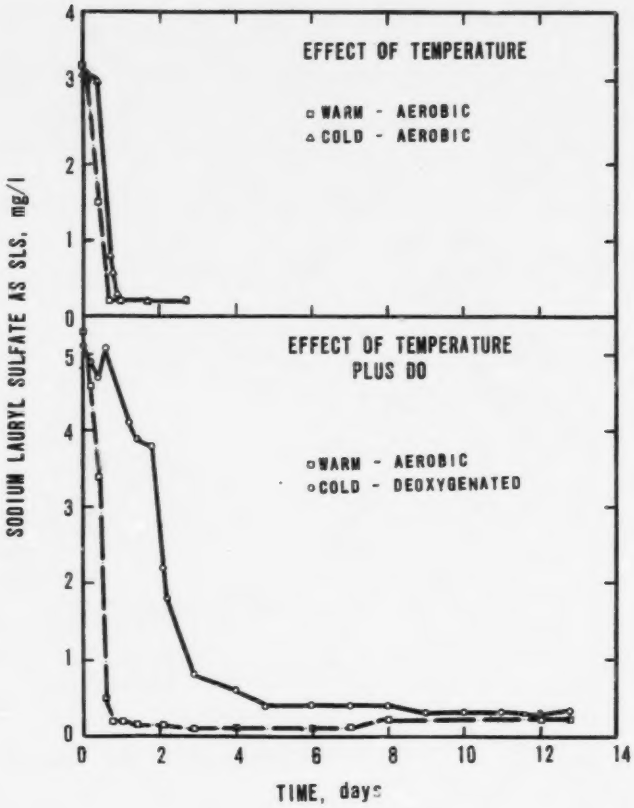

Figure 3. Behavior of sodium lauryl sulfate in a simulated stratified impoundment.

on the upper graph of Figure 3 show that temperature had little or no effect on the rate of degradation of the SLS. There was 94 percent degradation in less than 24 hours in the warm-aerobic environment and 94 percent in 24 hours in the cold-aerobic environment. The results in the warm-aerobic zone are typical of those found by other investigators in laboratory experiments.[7,20]

In Figure 3, comparison of the curve for cold-aerobic conditions (upper graph) with the curve for cold-deoxygenated conditions (lower graph) indicates that low dissolved oxygen had a significant retarding effect on the degradation of SLS compared to the effect of temperature.

Linear Alkylate Sulfonate Removal

The curves on the lower graph of Figure 4 show that in the warm-aerobic zone the LAS was 89 percent degraded in 29 days, and that approximately 84 percent was degraded in 5 days. The LAS in the cold-deoxygenated zone was degraded

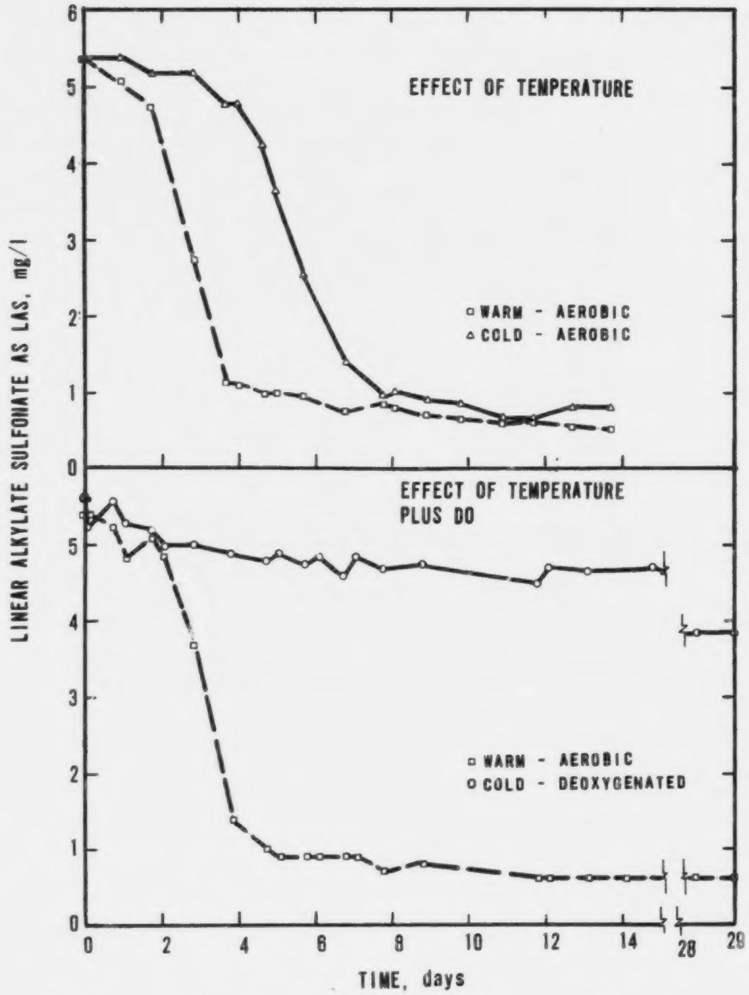

Figure 4. Behavior of lauryl alkylate sulfonate in a simulated stratified impoundment.

at a relatively uniform rate, reaching 32 percent in 29 days.

A temperature and oxygen effect test was also performed. The curves on the upper graph of Figure 4 show that cold caused the LAS to persist to the eighth day in contrast to warm temperature conditions where the LAS persisted approximately 4 days.

The effect of low dissolved oxygen is shown by comparing the cold-aerobic curve of the upper graph with the cold-deoxygenated curve of the lower graph. Under cold-deoxygenated conditions, the LAS persisted for 29 days with only a 32 percent degradation, while under cold-aerobic conditions the LAS was 82 percent degraded in 8 days. The results show that low dissolved oxygen had a more significant retarding effect on the rate of LAS degradation than did low temperature.

Although biodegradation was the cause of the major portion of SLS and LAS removal, adsorption, volatilization, or chemical hydrolysis may have caused the removal of some of the test material. Because SLS and LAS probably were not affected by volatilization and chemical hydrolysis under the conditions of this study, they were not considered possible mechanisms for the removal of these substances. The solids recovered during the material balance procedure showed that, in all cases, the sediment collected was greater than 70 percent of the theoretic amount available, as calculated by the decrease in TSS concentration during the test. The maximum amount of synthetic detergent extracted from these sediments was only 0.5 percent of the total amount of synthetic detergent added at the start of the test. Therefore, adsorption accounted for an insignificant portion of the SLS and LAS removal. This indicated that biologic degradation was the major mechanism of removal.

Pesticide Removal

Figure 5 shows that the concentration of 2,4-D remained relatively constant in the bacterially inactive control tank. The combined effect of low temperature and low dissolved oxygen, as shown on the lower graph of Figure 6 caused 2,4-D to persist for approximately 80 days in the cold-deoxygenated zone compared to only 6 days in the warm-aerobic zone. Duplication of these test conditions carried on simultaneously showed the same results in the

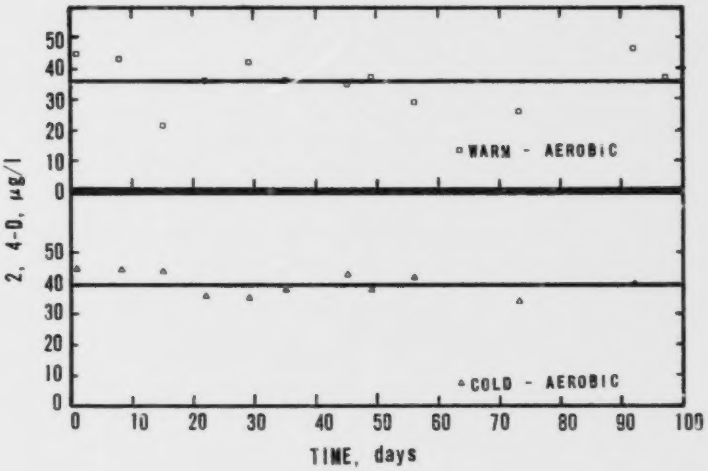

Figure 5. Behavior of 2,4-D in a bacterially inactive control.

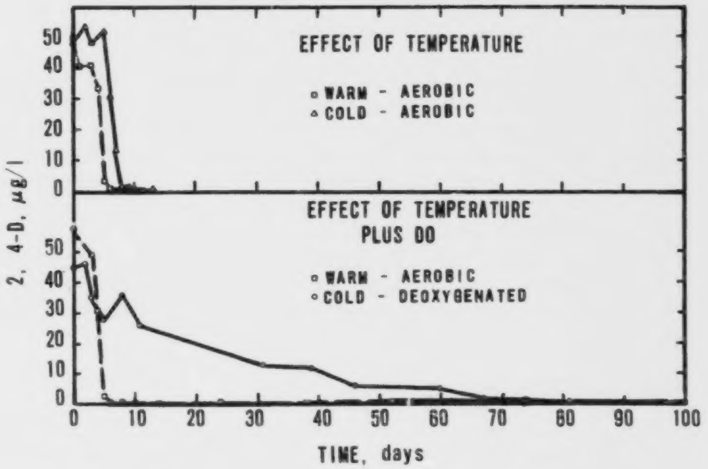

Figure 6. Behavior of 2,4-D in a bacterially active system.

warm-aerobic zone and persistence of approximately 55 days in the cold-deoxygenated zone. With each set of results, the order of magnitude of the difference between 2,4-D behavior in the warm-aerobic and cold-deoxygenated zones was large enough to indicate increased persistence because of the lower temperature and oxygen concentration.

The upper graph of Figure 6 shows that low temperature did not have a pronounced effect on the persistence of 2,4-D. For example, there was approximately 100 percent degradation in 6 days in the warm-aerobic zone and 8 days in the cold-aerobic zone.

Comparison of the cold-aerobic curve of the upper graph with the cold-deoxygenated curve of the lower graph shows that low dissolved oxygen was the major factor in causing the relatively long persistence of 2,4-D in a cold-deoxygenated environment. The excellent performance of the simulated stratified impoundments in duplicating results under identical test conditions is shown by comparing the warm-aerobic curves of the upper and lower graphs.

Figure 5 shows that 2,4-D was not removed to any great extent in the bacterially inactive control. When the lines shown on Figure 5 were used to calculate the amount of 2,4-D removed from the control, less than 1 percent of the initial amount was lost. Comparing this to the essentially complete removal under the bacterially active test condition shown in Figure 6 indicates that biologic degradation was the mechanism of 2,4-D removal in the river water tested.

The material balance showed that adsorption did not remove a significant amount of pesticide from either the control or natural river water. The amount of adsorbed pesticide recovered from the sediment collected during the material balance procedure was only 0.03 percent of the total weight of 2,4-D initially added, although at least 70 percent of the theoretic amount of suspended solids was recovered and extracted for 2,4-D. Therefore, biologic degradation (Figure 6) was concluded to be the mechanism of disappearance of 2,4-D.

DISCUSSION

Persistence of LAS and SLS

Under warm-aerobic conditions both LAS and SLS were readily biologically degraded. The SLS was 96 percent degraded within 24 hours and the LAS 84 percent in 5 days.

Under warm-aerobic conditions, LAS was 89 percent degraded during 29 days of testing. Results showed that LAS is more resistant to degradation than SLS.

The effect of temperature on the biologic degradation of LAS and SLS was observed by maintaining an aerobic condition in both a warm and cold environment. LAS was seen to persist approximately 4 days longer under lower temperature (12°C) than under higher temperature conditions (23°C), whereas SLS was only slightly affected by the lower temperature. Renn *et al.*[4] cite that neither the LAS- nor BOD-removal efficiency of an extended aeration activated sludge system for a small community declined during winter operation (13°C). Normal operating temperature was 23°C. The BOD and LAS removal was 80 to 90 percent.

Comparison of LAS and SLS results in the cold-aerobic and cold-deoxygenated environments showed that at 10°C the low dissolved oxygen caused both LAS and SLS to persist longer. The LAS showed a greater resistance to biologic degradation under adverse oxygen conditions. Klein and McGauhey[6] support this finding. They noticed in their investigation of septic tank systems that LAS was readily degraded under aerobic conditions in the percolation fields and only slightly degraded under anaerobic conditions of the septic tank.

Finally, a comparison of degradation of LAS and SLS in warm-aerobic and cold-deoxygenated environments shows that low temperature and low dissolved oxygen concentration increased the persistence of both substances. SLS was 92 percent degraded in 5 days instead of 96 percent in 24 hours, while LAS was degraded 32 percent instead of 84 percent in 29 days. These results support the suggestion by Wayman[7] that low temperature and anaerobic conditions inhibit the breakdown of both soft and hard detergents.

The data produced by the simulated stratified impoundments is in general agreement with laboratory and field investigations for synthetic detergents. Higher microbial populations present in the field investigation of an activated sludge treatment plant[4] may account for greater LAS removal efficiency. Also, the possibility of microbial adaptation to synthetic organics in an environment with a continual supply of the substrate may account for increased biologic degradation under field conditions.

Persistence of Pesticides

Similar to LAS and SLS, 2,4-D was most persistent under low temperature and low dissolved oxygen conditions (55-80 days) and least persistent under warm-aerobic conditions (6 days). Other investigators have shown that 2,4-D is microbiologically degraded in soils under aerobic[21-23] conditions and that anaerobic conditions inhibit degradation.[24,25] Persistence of 2,4-D in soils has been reported to vary from days to months.[26]

Although frequently studied in soil systems, there are few reports on the behavior of 2,4-D in water. Faust and Aly[2] found that 3 milligrams per liter of 2,4-D persisted for 120 days in aerated natural lake water even when seeded with 100 percent settled sewage by volume. Interpretation is difficult, as the conditions of their experiment and analytic techniques were not defined. Erne[27] reported that 8 milligrams per liter of 2,4-D in ditch water had decreased by only 0.8 milligram per liter in 170 days and that 1 milligram per liter of 2,4-D in well water decreased by 0.08 milligram per liter for the same period. The 2,4-D concentration in these studies were 20 to 160 times as great as in the study reported in this paper.

Whiteside and Alexander[23] reported that 2,4-D is more readily degraded after a system has been exposed to the substance and given time to acclimate. The experiments described in this paper utilized a natural river water, however, and no attempt was made to acclimate the microorganisms in the systems to 2,4-D.

Variable Behavior

The studies reported here indicate that oxygen concentration and temperature have an effect on 2,4-D biodegradation under various conditions in a common apparatus. 2,4-D has not been investigated, however, in a sufficient number of various water environments to determine its general behavior. A comparison of the two studies of 2,4-D[2,27] and the one reported in this paper illustrates the variable behavior of 2,4-D in water.

There are at least two reasons for this variation. The initial concentrations, from 50 micrograms per liter to 8 milligrams per liter in the three studies, may have been a cause. Variation in types of water (river, lake, lake plus sewage, ditch, and well) may also have been a factor. This is supported by soil investigations that show that microbiologic acclimatization is involved in the biologic degradation of 2,4-D.[25] Therefore, 2,4-D might initially persist in a water where acclimatization is possible. Bacterial counts were not given in the other reports,[2,27] and therefore, the results are difficult to interpret.

FUTURE PLANS

Lindane and endrin, also studied with the same apparatus showed a decline in concentration during the 100-day test period. These data were not reported because the mechanism of disappearance could not be determined with certainty. This phenomenon should be investigated further. Also, field studies of stratified reservoirs, particularly those used for water supply, should be conducted to test for synthetic organic persistence.

SUMMARY AND CONCLUSIONS

Laboratory and field investigations of the persistence of synthetic organics in a water environment and in stratified impoundments have been neglected. The success of streamflow regulation projects in enhancing water quality depends upon the availability of knowledge defining substances that will degrade and persist under natural impoundment conditions. This paper has shown qualitatively the behavior of a range of synthetic organics in simulated stratified impoundments, using a natural river water. The results indicate that particular environments that may be present in impoundments could inhibit the degradation of certain synthetic organics. This could then lead to an eventual deterioration of water quality in a system planned for enhancement of downstream water. Under the conditions of this study the following conclusions can be made.

1. Sodium lauryl sulfate (SLS), linear alkylate sulfonate (LAS), and 2,4-dichlorophenoxyacetic acid (2,4-D) were biologically degraded under warm-aerobic, cold-aerobic, and cold-deoxygenated conditions.

2. Low temperature reduced the rate of biodegradation of SLS, LAS, and 2,4-D.

3. Low oxygen concentration reduced the rate of biodegradation of SLS, LAS, and 2,4-D more than low temperature.

4. The greatest resistance to biodegradation for SLS, LAS, and 2,4-D was caused by the combined effect of low temperature and low dissolved oxygen concentration.

ACKNOWLEDGMENTS

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Schwartz, H. G., Jr. Microbial degradation of pesticides in aqueous solutions. JWPCF. 39:1701-14. Oct. 1967.

PART III

ENGINEERING
RESEARCH
PAPERS

PAPER NO. 7

IMPOUNDMENT DESTRATIFICATION BY MECHANICAL PUMPING

W.H. Irwin, J.M. Symons, and G.G. Robeck

This paper, the first in a series of eight on artificial destratification for reservoir water quality control, discusses the data obtained during the summer of 1964. During that summer a raft-mounted mechanical pump was placed in four small Southern Ohio lakes and the effect of pumping measured on the profiles of temperature, pH, and concentrations of dissolved oxygen and carbon dioxide. Four discussions and a closure are also reprinted.

The correct reference citations are:

Irwin, W. H., J. M. Symons, and G. G. Robeck. "Impoundment Destratification by Mechanical Pumping." Journal Sanitary Engineering Division, Proceedings of the American Society of Civil Engineers, 92, SA6, 21-40. December 1966.

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INTRODUCTION

Since 1961 Federal construction agencies of the United States Government have been required, by law, to consider the regulation of streamflow as a method of controlling water quality where pollution cannot be prevented by treatment. Under Public Law 660, as amended, these agencies must consider the inclusion of storage capacity in their new impoundments for the purposes of streamflow regulation. The United States Department of Health, Education, and Welfare was assigned the responsibility of performing research necessary to evaluate any proposed streamflow regulation project that had water quality control as its objective.

Because impoundments will be the source of water in most streamflow regulation schemes, good quality water must be available at all times if augmentation is to be successful. Research must be conducted, therefore, on the improvement of dam design for the maintenance of good quality impounded water and on the development of methods to predict and control water quality in existing and proposed impoundments. Previous papers[1-4] have reported on other phases of the research work in this general area performed by the project staff.

Since thermal or density stratification often causes a general deterioration of water quality in the deeper areas of impoundments, and controlled releases are often drawn from these lower levels through power turbines, a deterioration, rather than the desired improvement, in downstream water quality is often the result. If, however, sufficient mechanical energy is applied to an impoundment to break the thermal stratification pattern, good quality water can be maintained throughout the entire water mass. Therefore, destratification would generally benefit impoundment water users as well as released water users. The purpose of this paper is to examine research done in this area and to present detailed results of destratification of impoundments by mechanical pumping.

Thermal Stratification

Stratification is widespread, occurring at all latitudes in the United States. Lakes are often stratified thermally and chemically during spring, summer, and autumn, with the cold water trapped in the lower areas. Thermal stratification usually occurs first. Cold water has a greater density than warm water. Because lake waters are heated from above, the upper water decreases in density, but the deeper water remains cool and retains a greater density, thereby forming density layers. A wind flowing across the surface of a lake stirs the water to a depth commensurate with the density resistance. The velocity, frequency, and duration of the wind, the shape and size of the impoundment basin, and the direction of the long axis of the reservoir in relation to the prevailing winds are factors influencing impoundment mixing. Deeper stirring is accomplished by winds that are unobstructed and persistent.

When thermal stratification begins, a rapid change in density may occur 6 to 10 feet below the water surface. If turbulence from winds occurs, the zone of rapid density change may be lowered to 30, 40, or 60 feet. Each windstorm is likely to lower the zone of density change even though stratified layers remain. In certain instances, however, a strong storm might make an impoundment nearly isothermal.

As the air and water are cooled in autumn, the upper water becomes more dense and sinks to a level appropriate for its density. This process continues until the entire lake has a uniform density. This gradual mixing is termed natural ''impoundment overturn.''

Quality Considerations

In a thermally stratified situation, the lighter warm water forms a cover over the denser bottom waters and prevents their receiving atmospheric reaeration. The dissolved oxygen (DO) concentration is generally reduced to zero because of the presence of oxygen-demanding materials.

Under anaerobic conditions, changes occur that tend to lower water quality. Hydrogen sulfide as well as the sulfides of iron and manganese appear and create a negative oxidation-reduction potential. In addition, soluble iron and soluble manganese concentrations may increase in the bottom water. The pH will decline, and the carbon dioxide concentration will increase. Anaerobic decomposition can produce some undesirable organics, undesirable because of tastes and odors and, occasionally, because of toxicity. Even the cold temperature of these bottom waters might be considered a pollutant because this characteristic might adversely affect the downstream use.

Water of this poor quality could well cause a deterioration of downstream quality instead of the benefit that controlled releases are supposed to create. The purpose of artificial destratification is to prevent this quality deterioration and thereby to provide a continuing source of good quality water for streamflow regulation or water supply. Other methods of quality improvement are available, such as reaeration after release, but these have the disadvantages that they only raise the DO concentration, do little to the other quality parameters, and do not improve the water quality in the impoundment itself.

LITERATURE REVIEW

Until recently few attempts have been made to artificially control thermal stratification. Hooper, Ball, and Tanner[5] and Grim[6] have made efforts to break stratification by mixing reservoir waters with pumps. Hooper *et al.*[5] pumped cold water from the bottom to the top of the lake, and Grim[6] pumped the warm water from the top to the bottom. Neither attempt was carried to complete destratification, but both showed progress. Water jets were installed in the walls of a storage reservoir by Cooley and Harris[7] to produce currents and prevent the formation of thermal stratification.

Several investigators have attempted artificial destratification by injecting diffused air into an impoundment. Derby[8] attempted this in 1953 in Lower Hollywood and Encino reservoirs. He stated that "Insufficient horizontal circulation prevented complete success, but the thermocline was definitely lowered in both reservoirs." Riddick[9] circulated the water of a small reservoir by releasing air 8 feet below the surface. The concentrations of carbon dioxide, color, phytoplankton, and iron decreased, but zooplankton and carbonate alkalinity concentrations increased. Schmitz and Hasler[10] released compressed air near the bottom in an attempt to stir two small lakes. The authors stated that the thermocline continued in one. Nickerson[11] described the stirring of a small reservoir by the use of an airblower similar to those used in aerated sewage lagoons.

Ford[12] and Koberg and Ford[13] described the continuous mixing of lake Wohlford (2,500 acre-ft) by application of compressed air. Ford used a 210-cfm compressor powered by a 50-horsepower electric motor to apply the air through a 1-1/2 inch plastic pipe supported approximately 5 feet from the bottom of the lake. The air was applied over a linear distance of 60 feet. Compressor operation of approximately 75 hours over a 7-day period was sufficient to destratify this body of water, and the lake was maintained in a destratified condition from April throughout the summer by intermittent air injection.

Several authors have described destratification through the use of a device manufactured by Aero-Hydraulics, Limited.[14-17] The Aero-Hydraulics Gun, popularly known as the ''Bubble-Gun,'' is a low-head high-volume positive displacement pump, which uses periodic air bubbles rising in a polyethylene tube to promote piston action. The company has had several successful installations of the equipment, and this device has been reported to transfer ''...up to 5 million gallons of water per day from bottom to surface-with a power input of less than 1 hp.''[17] The installation of these devices is well summarized by Bryan.[18]

Only one reference, Ridley,[19] was found that proposed a device similar to that used in the studies described below. Ridley proposed to control the quantity of good quality water in the King George VI Reservoir by using an electrically-driven axial-flow pump mounted on a raft and capable of transferring up to 40 mgd of bottom water. As of December 1965 this project was still under way, and the results are not known.

A complete bibliography on the subject of artificial destratification is given at the end of Part III - Engineering Research Papers.

METHOD OF DESTRATIFICATION

Cold, dense water was pumped from near the lake bottom to the surface, where it could mix with the warm water, lose considerable density, and absorb oxygen. This water, after its discharge at the surface of the lake, sank to the level of its new density.[20] Because the water was not lifted above the lake's surface, the only energy required was that needed to lift the difference in water density, overcome the inertia of standing water, create a velocity through the pipe and at the exhaust, and overcome the hydraulic head losses in the mechanical system.

The density of fresh water at 4°C is 1.0 gram per cubic centimeter and at 30°C, 0.995675. The weight to be lifted, therefore, is 0.004325 gram per cubic centimeter, or only 0.43 percent of the weight of the water. At the above temperatures, the weight to be transferred would be only 18 tons per million gallons (approximately 4,200 tons). Because the water is seldom as cold as 4°C at the bottom or as warm as 30°C at the top, the percentage of weight to be lifted would be less for most impoundments.

The physical nature of density stratification in impoundments theoretically makes possible removal of all the cold layers of water in a lake from one location if the pump is located at the deepest part of the basin. Water leaving the pump outlet would have a considerable velocity and would produce a current across the lake. Because this discharge water does not return to the bottom of the reservoir, an equal volume of bottom water must be drawn into the pipe opening near the bottom of the lake. These upper and lower water currents then would promote mixing of the entire body of water even though the pump was placed at a single location (Figure 1).

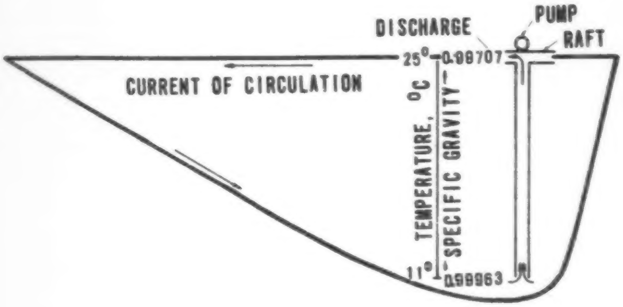

Figure 1. Density variation and destratification currents.

To circulate the cold water, a mixed-flow pump with a pipe attached below the impeller and extending downward to just above the lake bottom, was mounted on a float (Figure 2) anchored over the deep part of the lake. The lower end of the extension pipe was equipped with a strainer and a sheet-metal plate that admitted water from the side to avoid circulating the bottom mud and ooze. The 12-inch-diameter mixed-flow pump was powered by a 16-horsepower gasoline engine and operated at 2,000 rpm. At this speed, the output of the gasoline engine was approximately 12 horsepower. Pumping against a low elevation head, the pump had a capacity of 13 acre-feet per day. In other capacity units, this is 4.2 mgd, 6.4 cfs, or 2,880 gpm. The velocity of the water discharging from the pump was approximately 8.4 fps. As shown in Figure 2, the pump discharged water just at the impoundment surface through a short straight section. This discharge created sufficient turbulence to mix the cold bottom water with large quantities of warm surface water. The transfer of heat changed the density and prevented the pumped water from sinking back to the bottom. The first color photograph shows the pump being lowered into position and gives

an idea of its size. The second color photograph shows the pump in operation. Note the turbulent discharge at the surface.

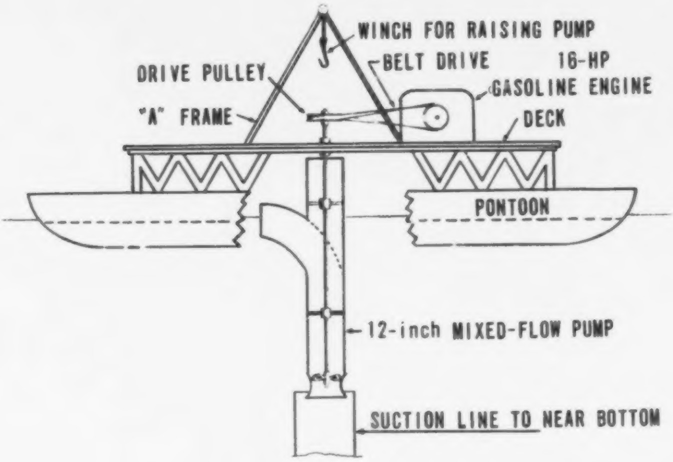
WINCH FOR RAISING PUMP
DRIVE PULLEY
BELT DRIVE
16-HP
GASOLINE ENGINE
"A" FRAME
DECK
PONTOON
12-inch MIXED-FLOW PUMP
SUCTION LINE TO NEAR BOTTOM

Figure 2. Design of pump, pontoon raft, and engine for destratification.

IMPOUNDMENTS STUDIED AND RESULTS

The apparatus was installed in several small lakes in eastern Ohio (Figure 3). All lakes stirred in 1964 were well stratified, thermally and chemically. Because stratification was permitted to develop before pumping was begun, the changes observed are representative of the process of breaking up stratification. The small size of the reservoirs permitted observations on a number of varying situations rather than on only one body of water. The information acquired during the early studies could then be adapted and applied to those reservoirs pumped subsequently. The studies on the three small lakes also aided in establishing endpoints of mixing and revealed factors that were not recognized as important at the beginning of the investigation.

The reservoirs pumped in 1964 were formed by the construction of earthern dams among steep hills; they had

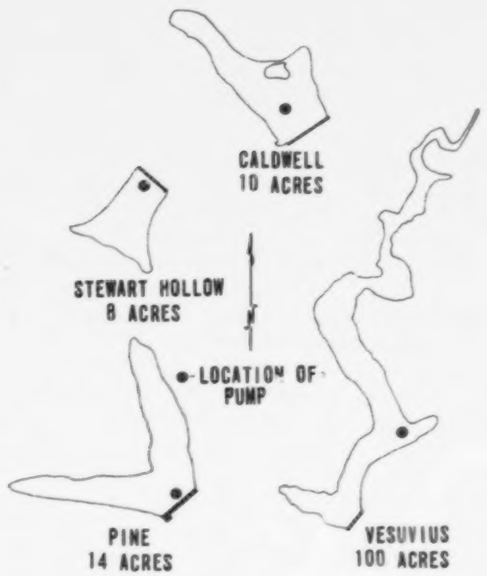

Figure 3. Lakes mixed.

small watersheds and were well protected from the wind, both by hills and trees. The soil was poor forest land with many stones, including sandstone that showed traces of iron. The water fertility was derived from debris of the forest floor, the soil of the watershed, and the accumulated organic material in the lake. Because of the protection from winds and the comparatively steep basin walls, the waters of these impoundments stratify thermally and chemically in early spring and remain stratified until heat is lost from the upper layers in the fall at the onset of cold weather.

Temperature and concentrations of hydrogen ions (pH), DO, and dissolved carbon dioxide were determined from top to bottom for each lake before and after pumping. In addition, temperature measurements were recorded at frequent intervals to observe the progress of thermal destratification. Finally, the above-noted parameters were remeasured for each lake, some time after pumping had ceased, to learn the permanency or variation of conditions. Also, some water samples were collected for analyses of algae and concentrations of nitrogen, sulfide, and phosphorus. Most of the samples were taken at the pump site, except for the temperature profiles, which were taken at several places.

Temperatures were taken with a Yellow Springs Instrument Co. resistance thermometer. Dissolved oxygen samples were collected with a DO sampler as detailed in ''Standard Methods for the Examination of Water and Wastewater,''[21] and the DO concentration determined in the field by the azide modification of the iodometric method. The pH was determined colorimetrically by use of a pH indicator and a standard set of Taylor tubes. The carbon dioxide concentration was determined in the field by the titrimetric method for free carbon dioxide. Samples for the latter two analyses, as well as those mentioned at the end of the previous paragraph, were taken from the excess water in the DO sampler.

Stewart Hollow Lake

Stewart Hollow Lake, Ross County, Ohio, with a surface area of 8 acres, a maximum depth of 25 feet, an average depth of about 15 feet, and a computed volume of 120 acre-feet was mixed by pumping for 20.5 hours between July 20 and 23. Pumping times are always shorter than calendar times because of occasional shutdowns. Figure 4 contains graphs representing the temperature, DO, pH, and carbon dioxide profiles as they existed before and after the first pumping. (All data for these and other parameter changes were taken at the pump site.) Each parameter became more nearly uniform in concentration with depth as mixing progressed. Temperature data showed that this occurred not only at the pump site, but also throughout the entire lake. A few additional hours of pumping probably would have made the water density and water quality even more uniform with depth, but pumping was stopped, and the machine moved to other lakes. Later, on September 1, the pump was returned to Stewart Hollow Lake for 13.5 hours additional pumping because the water was restratifying. Figure 5 contains profiles of four water quality parameters at the beginning and end of the second pumping. Again, stratification was largely eliminated.

The second pumping was completed on September 2, and the lake conditions were remeasured 3 weeks later, on September 23. These data are shown in Figure 6. As shown by the temperature data, the water near the bottom cooled, probably because of a loss of heat to the cooler mud. Algal

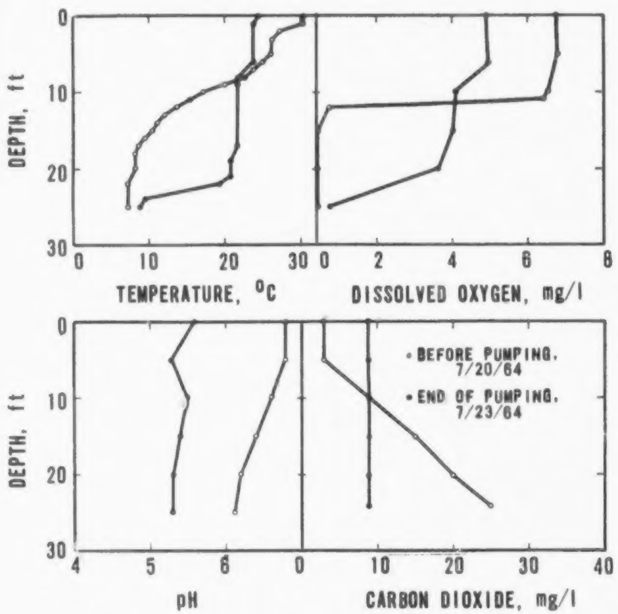

Figure 4. Parameter changes in Stewart Hollow Lake, first mixing.

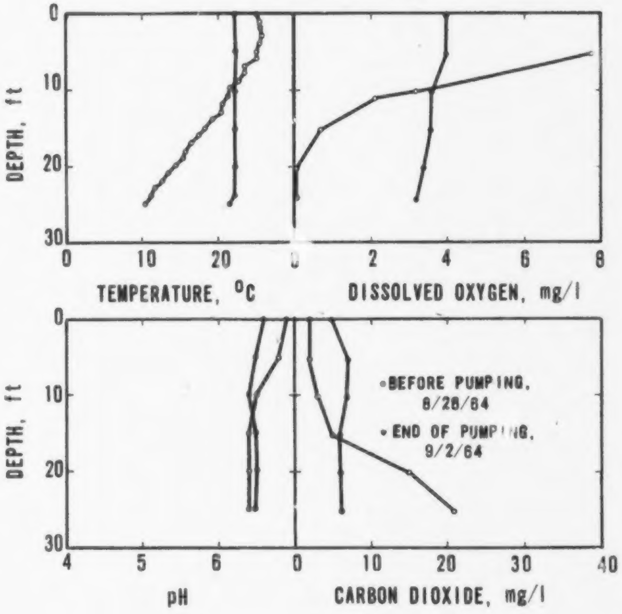

Figure 5. Parameter changes in Stewart Hollow Lake, second mixing.

action plus atmospheric reaeration caused the DO concentration near the surface to increase, but a continuing oxygen demand in the lower waters caused a decline in the oxygen concentration below 10 feet. The pH and carbon dioxide concentration did not change significantly. Finally, the usual fall bloom of blue-green algae did not occur in Stewart Hollow Lake in 1964.

Caldwell Lake

Caldwell Lake, Ross County, Ohio, has a surface area of 10 acres, a maximum depth of 20 feet, an average depth of of about 10 feet, and a volume of 100 acre-feet of water. The lake was pumped during July 29 and 30, 1964. Figure 7 shows the water quality characteristics before and at the end of the pumping. Actual pumping was conducted for only 8 hours. At this time the temperature, as shown in Figure 7, was so nearly uniform that pumping was stopped. Although a few more pumping hours probably would have made more progress, at this time the lake was nearly homogeneous. Figure 8 shows the profiles of the water quality parameters of Caldwell Lake, 7 weeks later, on September 23. In the fall of 1964, the usual bloom of blue-green algae was absent from Caldwell Lake.

Pine Lake

Pine Lake, Ross County, Ohio, has a surface area of 14 acres, a total depth of 17 feet, an average depth of 7 feet, and a volume of 98 acre-feet. The pump was in operation for 35 hours between August 5 and 7. Temperature changes recorded during the pumping showed that destratification was progressing slowly. At completion, when the suction pipe was lifted, a rupture was found that delayed mixing by removing lake water from 9 feet below the surface, rather than from the intended 16 foot depth. Figure 9 shows the impoundment conditions before and at the end of pumping. Despite the accident, this lake was nearly destratified by pumping. Water quality profiles were measured again 6 weeks later, on September 23. Figure 10 shows the profiles taken at the end of pumping and 6 weeks later. The little cooling and chemical restratification detected was not nearly so pronounced as in Stewart Hollow or Caldwell Lake. Like Stewart Hollow and Caldwell Lake, however, the usual fall blue-green algal bloom did not occur.

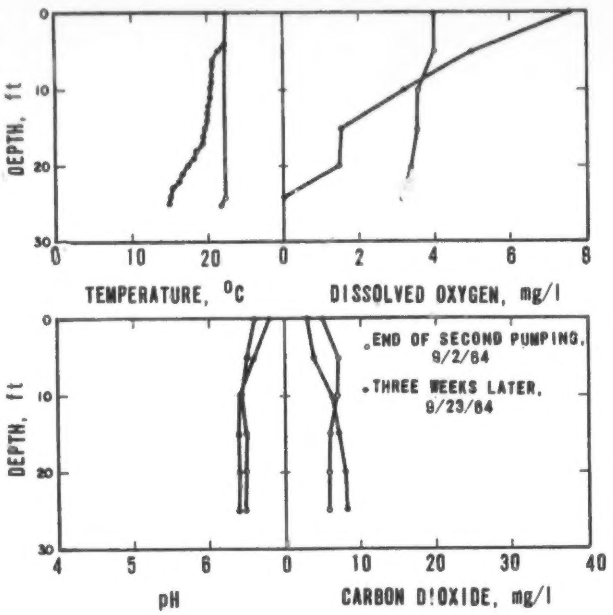

Figure 6. Parameter changes in Stewart Hollow Lake, after second mixing.

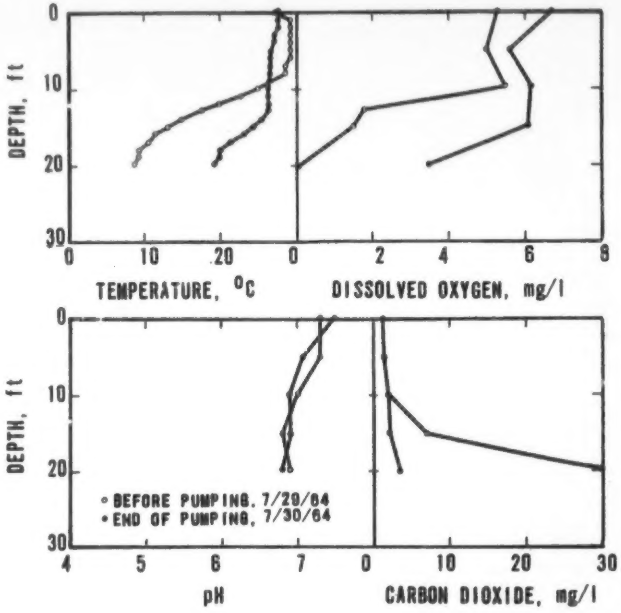

Figure 7. Parameter changes in Caldwell Lake, during mixing.

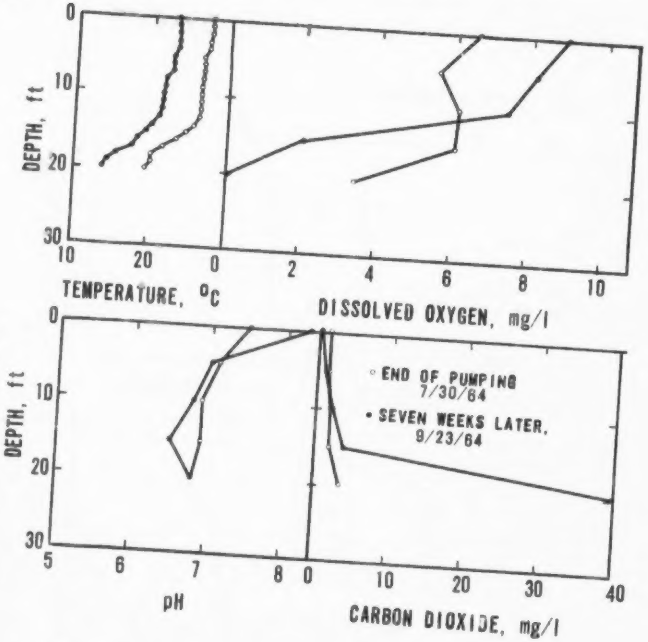

Figure 8. Parameter changes in Caldwell Lake, after mixing.

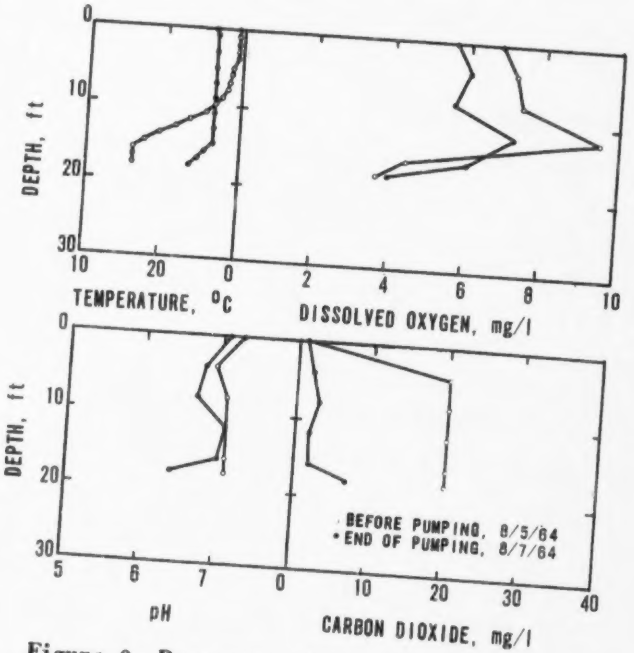

Figure 9. Parameter changes in Pine Lake, during mixing, water accidentally drawn from nine foot depth.

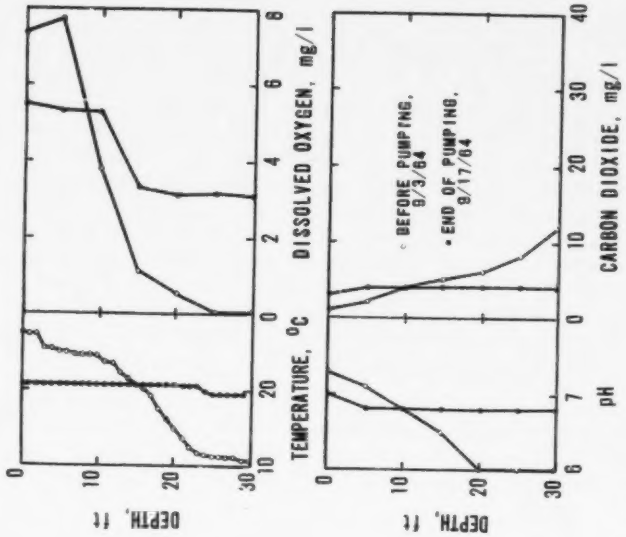

Figure 11. Parameter changes in Vesuvius Lake, during mixing.

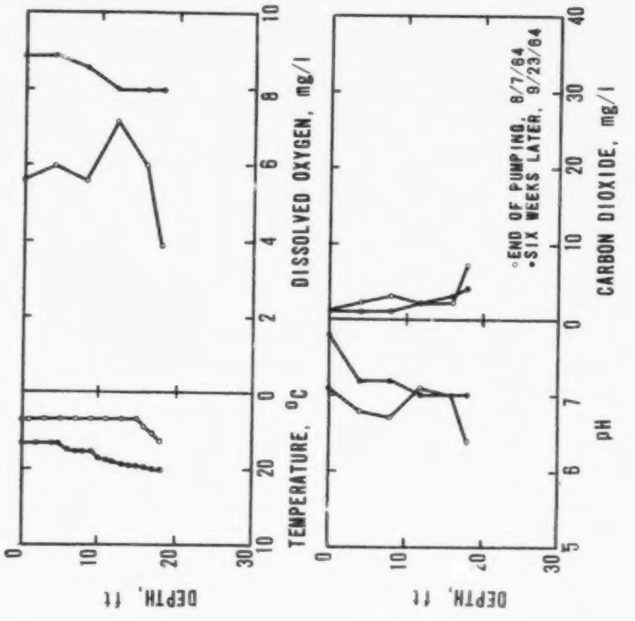

Figure 10. Parameter changes in Pine Lake, after mixing.

Vesuvius Lake

With three small lakes successfully destratified, the equipment was moved to a larger impoundment in Lawrence County, Ohio. Vesuvius Lake has a surface area of 100 acres, a total depth of 30 feet, an estimated average depth of 13 feet, and a computed volume of 1,260 acre-feet. The reservoir is long and narrow, no wider than 800 feet and extending among the hills about 3 miles. The lake was pumped 208 hours between September 3 and 17. Figure 11 shows the profiles for temperature and concentrations of DO, hydrogen ions (pH), and carbon dioxide at the beginning and end of pumping operations. Final records of the parameters 1 week later, September 23, in comparison to those of September 17 are shown in Figure 12.

Although Vesuvius Lake was more than ten times larger than any impoundment previously studied, the data obtained were remarkably similar to those previously presented. In general, the lake was mixed throughout its depth as shown in Figure 11, and there was a continuing loss of heat to the cold bottom and a tendency for the DO concentration in the lower waters to decline with time after pumping ended, as shown in Figure 12.

As with the other three lakes, which were mechanically pumped, the usual fall bloom of blue-green algae did not occur in 1964 in Vesuvius Lake. The additional water quality data taken in Vesuvius Lake were not as extensive or as complete as desired, but information in Table 1 shows the effect of mixing. The concentrations of both ammonia nitrogen and sulfide were originally highest near the bottom of the lake, but both parameters showed uniform distribution after mixing. This again confirms that the lake was thoroughly mixed with respect to depth.

Table 1. VESUVIUS LAKE WATER QUALITY DATA

Depth, feet	NH_3 - N, milligrams per liter		Sulfide - $S^=$, milligrams per liter	
	Before	After	Before	After
0	0.10	0.10	< 0.03	0.03
5	0.00	0.10	< 0.03	0.05
10	0.10	0.10	< 0.03	0.05
15	0.05	0.10	< 0.03	0.04
20	0.10	0.10	0.08	0.05
25	0.60	0.10	0.43	0.06

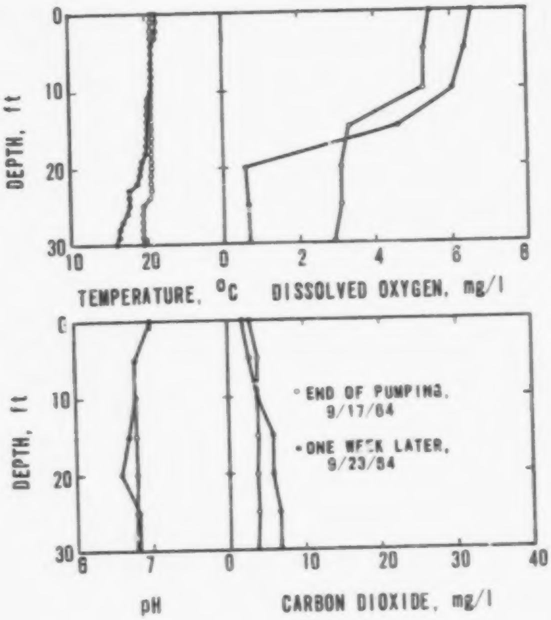

Figure 12. Parameter changes in Vesuvius Lake, after mixing.

Twice during the pumping operation, temperature profiles were taken at three points other than the pumping site. These points were distributed throughout the entire length of the lake. A study of the progressive change in the temperature profiles obtained from the various stations in the lake showed that the entire water mass was behaving as a unit (see Figures 13 and 14); that is, the temperature was the same at any given depth at any time during the pumping operation throughout the entire lake. This confirms the theoretical water movement postulated in Figure 1 and indicates that, if a lake is originally thermally stratified, all of the cold water drains to a central point and can be pumped out without moving the pumping equipment.

Using the depth-volume relationship for Vesuvius Lake and the profile of DO concentration before and after pumping, calculations can be made to show the changes in the total oxygen content in the impoundment during the

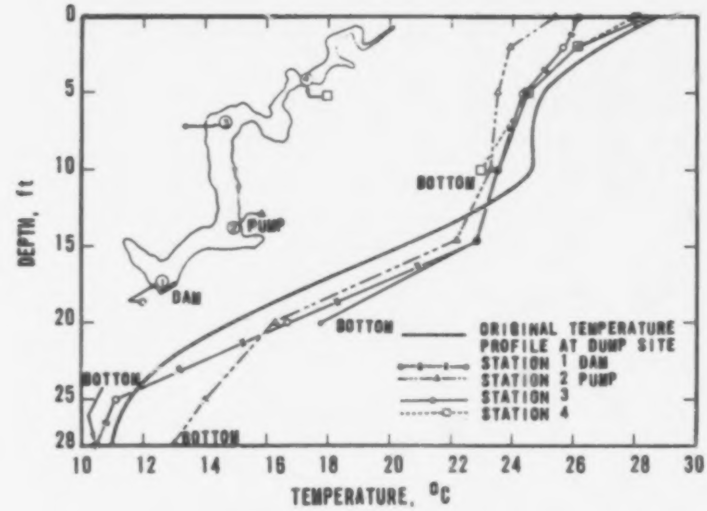

Figure 13. Temperature profiles at various stations in Vesuvius Lake after 2.5 days of mixing.

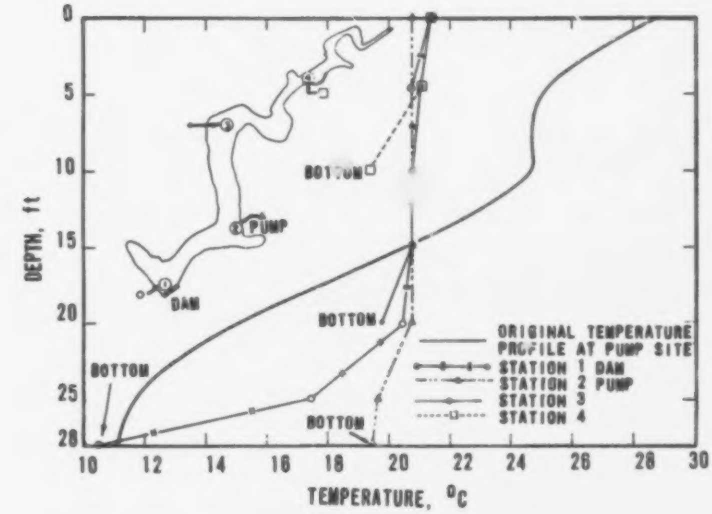

Figure 14. Temperature profiles at various stations in Vesuvius Lake after 8.5 days of mixing.

investigation. Before pumping began, there was about 15,200 pounds of DO in the entire water mass. At the completion of destratification approximately 15,000 pounds of oxygen was dissolved in the water. Because the total quantity of DO was essentially the same before and after pumping, the rate of oxygen transfer created by the mixing operation must have been about equal to the consumption of oxygen caused by bringing oxygen-demanding water from the lower zones of the lake to the surface.

DISCUSSION

Economics of Mechanical Pumping

Any destratification method should be reasonable in cost, easy to install, easy to operate, and capable of producing good quality water. Mechanical pumping with an axial-flow pump presents one practical approach to destratification because large capacity axial-flow pumps are available. Some summary data are contained in Table 2, which will be of help in judging the practicality of the pumping operations reported herein. The cost and the hours pumped vary from lake to lake because of the variations in volume and in percentage of cold water moved. In addition, Stewart Hollow Lake was pumped twice, Pine Lake was overpumped because of a rupture in a pipe, and Caldwell Lake was slightly underpumped. Probably the data for Vesuvius Lake are the best to date (1964), because the volume was high, the method functioned well, pumping time was relatively short, and results were good. Even here, however, the data (Table 2) for gasoline costs (Column 8,

Table 2. SUMMARY OF LAKE VOLUMES AND PUMPING DATA[a]

Lake	Surface, acres (1)	Total depth, feet (2)	Estimated average depth, feet (3)	Volume, acre-ft (4)	Volume cold water pumped, acre-ft (5)	Hours pumped (6)	Percentage cold water pumped[d] (7)	Gasoline cost, dollar total (8)	Input hp-hr (9)	Input hp-hr per acre-ft cold water pumped (10)	Input hp-hr per acre-ft lake volume[e] (11)
Stewart Hollow[b]	8	24	15	120	20.25	37.5	22.5	15.00	450	22.2	3.75
Caldwell	10	20	10	100	4.33	8.0	5.7	3.20	96	22.2	0.96
Pine[c]	14	17	7	98	8.9	35.0	25.5	14.00	420	38.3	4.3
Vesuvius	100	30	13	1300	112.3	208	11.2	82.00	2496	22.2	1.92

[a] Pump capacity, 13 acre-ft per day; Power input, 12 hp.

[b] Lake pumped twice.

[c] Lake overpumped (pipe broken).

[d] Based on original temperature profile.

[e] This figure depends on efficiency of mechanical equipment and percentage of water actually pumped in any given impoundment.

Table 2) and for horsepower-hours per acre-foot of lake volume (Column 11, Table 2) must be viewed with some reservation because only 11.2 percent of the lake volume had to be passed through the pump in order to mix the entire lake. In another situation, if the percentage of lake volume that has to be pumped to obtain complete mixing were higher, the total energy input per unit of lake volume would be correspondingly higher. Therefore, the generalization that approximately 2 horsepower-hours per acre-foot of lake volume is sufficient to mix all impoundments cannot be made.

Even the value (Column 10, Table 2) for horsepower-hours per acre-foot of cold water pumped must be interpreted with some reservations. The velocity through the device of about 8.4 fps created a velocity head of about 1.1 feet. Because a large part of the 12-horsepower input of the gasoline engine was needed to overcome the hydraulic losses inherent in the system the pump was hydraulically inefficient, and the data, represented in Table 2, therefore, are high.

One other feature of the data is worthy of note. A comparison of the volume of cold water that actually passed through the pump to the volume of cold water that initially existed in the impoundment shows that more water was moved than pumped. The pump's high discharge velocity, therefore, did create some beneficial effect by establishing currents in the lake. The importance of these currents must be further investigated in the future.

Influence on Quality

This paper reports on the first year (1964) of operation of a long-range project to investigate the potential of artificially modifying the density profile of impoundments. The data clearly show that bringing bottom water to the surface with a mixed-flow pump successfully destratified the four lakes that were studied. The mixing operation made the water quality parameters that were measured uniform in concentration from the surface to the bottom, at the pump site, and from one location in the lake to another. Temperature, as shown by temperature data and, therefore, density, pH, concentrations of DO and carbon dioxide, and plankton densities were made nearly uniform in the impoundments. Examination of the types of plankton showed that the usual fall bloom of blue-green algae did not occur in 1964 in any of these lakes, although the total number of algae did not decline. Because the changes noted above occurred in the absence of any unusual meteorological conditions, such as severe storms or unusually cold weather, they can be attributed to the operation of the pumping device.

In addition to the obvious influence that artificial destratification had on the water quality of the impoundments studied, several other interesting features of this operation can be extracted from the data. First, there was a definite tendency for Stewart Hollow, Caldwell, and Vesuvius to restratify thermally. This thermal restratification gave rise to chemical restratification as shown by the change in pH and concentrations of DO and carbon dioxide in the lower levels after pumping stopped.

The theoretical cause of thermal restratification is explained as follows: Because an impoundment changes from a uniform to a stratified condition in the spring under certain conditions of wind and terrain, it probably would do this again after a pumping operation because of the addition of heat by solar radiation and the loss of heat to the cold bottom muds. Changes in heat content in the surface and bottom waters produce a new temperature gradient, and unless winds are stronger than in spring, the gradient would tend to persist. Calculation of the heat gained or lost from the various layers of Vesuvius Lake during the mixing operation showed a net loss of about 13.8×10^9 BTU from the entire water mass. This would correspond to a decrease in the average temperature of the entire water mass of 2.3°C and demonstrates the above-mentioned heat loss to the cold bottom muds.

Although the general temperature of the water is higher after pumping than in the early spring, stable stratified conditions can be re-established because of the increase in the change of water density per degree of temperature as the water temperature increases. The density profile, not the temperature profile, in an impoundment controls stratification;[2] therefore, a modest temperature gradient at generally warm temperatures could produce stable density stratification. If this density gradient caused sufficiently severe chemical stratification to redevelop, an impoundment would have to be repumped during a summer season, at intervals controlled by the rate of chemical restratification, to maintain good quality water.

Second, calculations from the DO concentration data in Vesuvius Lake showed that there was no net increase in the total DO content of the lake during the pumping operation.

Although sufficient oxygen was supplied during pumping to meet the demand of the bottom water that was brought to the surface, the lake, at the end of the pumping operation, was left in an undersaturated situation with respect to DO. In general, however, periodic mixing, as noted above, probably would be required to maintain destratified conditions throughout a summer except for shallow lakes, such as Pine. This remixing would continually bring water that was undersaturated in DO to the surface where it could become reaerated through contact with the atmosphere. Although the net DO content of a body of water was changed little during the first mixing, as in Vesuvius Lake, the entire body of water, with time, should become adequately reaerated as repumping proceeds.

Third, an examination of the data shows that prevention of stratification rather than destratification is a better approach to impoundment management. Maintaining good quality water throughout an impoundment by starting the mixing in the spring would be more sensible than distributing bad quality water through the lake in the summer. As noted above, however, despite a spring destratification, most deep and protected impoundments would tend to restratify and, therefore, would have to be mixed again during the summer.

Finally, the rate at which an impoundment must be destratified is related to the rate of oxygen demand. If this rate is high, the mixing rate must be high, or DO will be removed from the water as fast or faster than it is added. In lower-demand-rate situations, however, the rate of mixing could be slower. For the conditions studied, mixing should be quite rapid, probably taking little more than 1 week to complete, if a decline in DO concentration throughout the water mass is to be prevented.

FUTURE PLANS

In August 1965, the apparatus used in the 1964 operations was placed in a lake approximately twice the size of Vesuvius Lake. Extensive sampling permitted a complete record of the changes in many water quality parameters during the pumping operations. The data are being evaluated and will be reported in a future paper.[22]

Plans for 1966 include an attempt to prevent the onset of thermal stratification through an early start, possibly in April, of the mixing operations. Also in 1966, the use of compressed air will be investigated as an alternative method for mixing a large body of water.

CONCLUSIONS

Although the data reported in this paper cover only the first year of a long-term project, several firm conclusions can be drawn at this time for application to destratification of impoundments of the sizes studied.

1. Mechanically pumping cold water from the bottom of a stratified impoundment and discharging it at the surface with a mixed-flow pump is an effective method of modifying the density profile. (The mixing of the four impoundments studied occurred in the absence of any external meteorological forces, such as severe storms.)

2. In all cases, the water quality of the entire water mass was improved, with respect to the parameters studied, by artificial destratification.

3. The usual fall bloom of blue-green algae did not occur in any of the four artificially destratified lakes, although the total number did not decline significantly.

4. The entire water mass of an impoundment can be mixed with the pumping apparatus located at one site, that is, over the deepest part of the reservoir.

5. Impoundments protected from the wind thermally restratify after pumping is completed.

6. The data shown in Table 2 concerning the horsepower-hours required to destratify a unit of impoundment volume can be used as a general guide, but will be influenced both by the mechanical losses in any destratification equipment and the quantity of cold water that exists in any given impoundment before a pumping operation is begun.

7. The rate at which a lake must be destratified is proportional to the rate of oxygen demand.

8. Only a part of the cold water moved from the bottom of an impoundment actually passes through the pump, the importance of water currents as an aid in destratification is thus indicated.

ACKNOWLEDGMENTS

The Ohio Department of Natural Resources through its Divisions of Wildlife and Forestry aided in the selection of usable lakes, and provided and helped to install some

of the equipment. Particular mention should be made of John Pelton, Clarence Clarke, Gary Apgear, and George Billy of the Section of Fisheries for their help and understanding of the problems involved. Cooperation was also enjoyed from the National Forestry personnel. The writers also wish to thank Glenn R. Gruber of the Engineering Activities staff for his help during these studies.

Discussion of

"IMPOUNDMENT DESTRATIFICATION BY MECHANICAL PUMPING"

Edward L. Thackston[*]

The data presented demonstrate that mechanical pumping is a workable method of mixing small or moderate-sized impoundments that have become density-stratified. Whether or not it is an economically feasible or desirable method is a question which cannot be answered at the present time. The data on work requirements presented in Table 2 are valuable in that this constitutes the first time that even order-of-magnitude figures on actual requirements for this method have appeared in the literature.

Although the unfortunate hydraulic inefficiency of the author's pump makes the actual values of required work unreliable, and possibly misleading to those who do not heed the authors' warnings, the true requirements may not be greatly different.

The high discharge velocity from the pump (8.4 fps), with its associated high velocity head and hydraulic losses, consume a considerable part of the pump's output. The substitution of a larger pump of the same capacity would obviously reduce this velocity head and thus reduce the input of power per unit volume actually pumped. However, at the same time, the reduction of the discharge velocity would reduce the momentum transferred from the discharge jet to the waters of the epilimnion, and, consequently reduce the mixing, or current-producing potential of the pump.

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Another possible disadvantage of lowering the exit velocity is that mixing of the cold hypolimnion waters with the warmer epilimnion would be inhibited. The resulting temperature of the discharge would tend to remain lower and the water might tend to drop back to the hypolimnion.

Therefore, although reduction of the discharge velocity would definitely reduce the power requirements for the installation described, the total work requirements would not be reduced proportionally. Because of the reduced mixing potential, to achieve complete mixing in the impoundment, the pump would probably have to operate longer, and a higher percentage of the hypolimnion would have to be passed through the pump. It is obvious from the figures in Column 7 of Table 2 that most of the water moved did not actually pass through the pump.

It is even possible that the total work required to break stratification might increase, rather than decrease, when a more efficient pump is substituted. The only way to resolve these questions is to try a more ''efficient'' pump, preferably on the same impoundments, and compare the resulting work requirements.

Because both mechanical pumping and compressed air mixing have now been demonstrated to be workable methods of mixing stratified impoundments, a determination of which is the more efficient (economical) would be of great value. Table 3 presents data from the literature on compressed air mixing in the same format as Table 2. Indian Brook Reservoir is from Riddick,[9] Lake Wohlford from Ford,[12] and Blelham Tarn from Bryan.[23] Riddick released air from a perforated pipe in the epilimnion; Ford released air from a perforated pipe in the hypolimnion, and Bryan released air from an Aero-Hydraulic Gun, or bubble gun, in the hypolimnion.

The most striking difference in values between compressed air mixing and mechanical pumping is in Column 9, work expended per unit of water pumped. The values for compressed air mixing are less than 10 percent of those for mechanical pumping. However, the values of work expended per unit volume of impoundment are about one-half those for mechanical pumping. This dramatically illustrates the value of the high discharge velocity of mechanical pumping in creating currents that help mixing.

For Wohlford and Blelham Tarn, where air was released directly into the hypolimnion, the total work required was

Table 3. PUMPING DATA FOR COMPRESSED AIR MIXING

Lake (1)	Surface, acres (2)	Depth, feet (3)	Average depth, feet (4)	Volume, acre-feet (5)	Volume cold water pumped, acre-feet (6)	Hours pumped (7)	Power input, horse-power-hours (8)	Input horse-power-hours per acre-foot pumped (9)	Input horse-power-hours per acre-foot of lake volume (10)
Indian Brook Reservoir	18	27.5		316	3,300	168	1,340	0.4	4.2
Wohlford	130	50	19	2,510		78	3,900		1.5
Blelham Tarn	27	44	21	575	150	335	~300	2.0	0.5

less than the 2 horsepower-hours per acre-foot of lake volume, which was the average for mechanical pumping. Whether this difference (signifying a greater efficiency for compressed air mixing) is real and significant cannot be known until more work has been done. A trial of both methods on the same impoundment, stratified to the same degree each time, is needed; or, possibly, a comparison of actual work requirements to the theoretical minimum could be made in each case. Until this is done, the question of mechanical pumping or compressed air will remain unanswered.

The use of artificial mixing to prevent, rather than to cure, stratification would seem to be one approach to lowering the associated costs. The main resistance to mixing is the stratification, or density difference, itself. If an impoundment is not stratified, its resistance to mixing is minimal. Conversely, if an impoundment remains in a mixed state, it cannot stratify. Thus, if mixing is begun before stratification, a high resistance to mixing should never develop and intermittent mixing will keep the impoundment unstratified. Total work input for the year should therefore be less than if the impoundment is allowed to stratify and be broken several times.

If this approach is followed, the method of air introduction near the surface used by Riddick[9] has several advantages, the most important of which is that the air does not have to be compressed to as great a pressure, and consequently power requirements are less. Another advantage is that the aerator can be float- or barge-mounted, and thus would be more flexible than a deeper, more permanent installation.

In Table 3, the high value of total work required for Indian Brook Reservoir (4.2 horsepower-hours per acre-foot of lake volume) was due to the fact that the air introduced did not raise the hypolimnion directly; it simply created a rolling current in the overlying epilimnion, which gradually skimmed off and entrained the hypolimnion. Thus, it is an inefficient way to break stratification. However, once stratification is broken, it should give more mixing for the money because, in the absence of density differences, most of the water in the rising plume will come from directly under the aerator, just as if the aerator had been on the bottom. It is even possible that continuous, or nearly continuous, high-level air release might be cheaper than intermittant low-level releases.

Whether or not the Aero-Hydraulic Guns are more efficient than perforated pipes, as is claimed, and as is indicated above, is another question that should be resolved eventually. The data are insufficient at present, but it is a possibility that must not be overlooked.

Because of the recent administrative decision of the Federal Water Pollution Control Administration that the discharge of low-oxygen-content water constitutes pollution, the necessity of solving some of the problems associated with stratification is more pressing. No practical suggestion should be overlooked as a possible solution. The work of the authors is a significant and valuable step in this direction and it is hoped that it will continue.

Discussion of

"IMPOUNDMENT DESTRATIFICATION BY MECHANICAL PUMPING"

Heinz Bernhardt[*]

Comparison of the instructive test results that were obtained after mixing various stratified impoundments using gasoline-powered pumps with the Wahnbachtalsperrenverband's results of aerating and mixing a medium-size impoundment using compressed air is interesting. The Wahnbach Reservoir, Siegburg, West Germany, has a volume of 33,740 acre-feet, a maximum depth near the dam of 141 feet, an average depth of 63 feet, and a surface area when full, of 530 acres. In 1964, the reservoir was aerated and mixed by blowing 210 cfm of air into the water at the maximum depth near the dam. Aeration began on June 11, 1964, after the reservoir had a marked stratification and a stability value for the entire reservoir of 400 kilowatt-hours. Aeration operations continued until November 11, 1964, without any interruptions. The reservoir was nearly completely destratified as early as the end of August. In September, a water temperature of 16°C at all depths was recorded throughout the lake, which is 3.4 miles long. The same positive results were achieved in this experiment as in the authors' study (particularly for Vesuvius Lake with respect to the oxygen supply for the oxygen demanding processes taking place at the bottom of the reservoir. A summary of the German publications containing results of aerating the Wahnbach Reservoir[24,25] will soon be published.[26]

Of particular interest here, however, is the comparison of the energy used to create the extensive mixing in the reservoirs mentioned, and the relationship of each amount of energy to the total content of the lake in question. The authors give an average value of 2 horsepower-hours per acre-foot (1.5 kilowatt-hours per acre-foot) of lake volume as a sufficient capacity to be installed for mixing a lake. They do, however, point out that this figure should be viewed with some reservations. For the Wahnbach Reservoir, the installed compressor capacity was 64 horsepower (47.5 kilowatts) (two electrically-powered compressors at the dam). They were in operation for approximately 1,440 hours to break down the stability of the lake to an extent similar to the reduction in Vesuvius Lake. After this time, the water temperature at the bottom was 15°C, at the top 16°C, with the exception of a deep narrow zone. In this deep part of the reservoir, the water at the bottom was one degree colder, so that the temperature drop here was 2°C. This temperature profile of the Wahnbach Reservoir may be compared to the stability condition of Vesuvius Lake after 8.5 days of pumping. Here too, at Station 1 near the dam (Figure 14) there was a larger difference in the temperature of the bottom and the top water, and at all other stations there was still a temperature difference of approximately 2°C after mixing.

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The amount of energy used in the Wahnbach Reservoir was 92,000 horsepower-hours (68,400 kilowatt-hours), 2.7 horsepower-hours per acre-foot (2.0 kilowatt-hours per acre-foot). An additional 49,000 horsepower-hours (36,500 kilowatt-hours) was required to reduce the stability to near zero. The relatively good agreement of this figure (2.7 horsepower-hours per acre-foot) with the average of 2 horsepower-hours per acre-foot given by the authors indicates an energy consumption of 2-3 horsepower-hours per acre-foot lake volume is perhaps justified as a rough approximation of general validity to break down most of the stability. The different morphological conditions of the various lakes will obviously influence this figure, but the Wahnbach Reservoir is considerably larger and deeper than Vesuvius Lake, as shown in Table 2. The comparison of energy consumption is particularly interesting because two different mechanical systems were used for destratification, pumps in the authors' experiment and the application of compressed air rising freely in the water in the studies described herein.

On the other hand, 1-25 kilowatt hours per acre per year for energy stated by Kittrell,[27] according to calculations made by Hutchinson, does not seem to have general validity. This can be deduced from Table 4.

Table 4. COMPARISON OF ENERGY REQUIREMENTS

Lake	Surface, acres	Total energy used, kilowatt-hours	Energy used per lake surface, kilowatt-hours per acre	Energy used per lake volume, kilowatt-hours per acre-foot
Stewart Hollow	8	336	42	2.8
Caldwell	10	72	7.2	0.72
Pine	14	313	22	3.1
Vesuvius	100	1,862	19	1.5
Wahnbach	530	68,400	129	2.0

Relating the energy to be used for extensive lake circulation to the volume of the lake produces approximately the same values for the four lakes examined in the paper, and for the Wahnbach Reservoir. If, however, the energy to be used is related to the surface of the lake, the values given in the paper are completely wrong for the larger Wahnbach Reservoir, which is also considerably deeper in relation to its surface. The reason for this is the different morphology of the Wahnbach Reservoir compared to the four lakes examined in the paper. Whereas their average depths are practically the same, the average depth of 63 feet for the Wahnbach Reservoir is considerably greater. Therefore, a rough estimate of energy requirements cannot be made based on lake surface; instead, lake volume should be used.

Even better estimates are achieved, if the energy requirements are related to the stability of the lake during the summer months. Symons, Irwin, and Robeck[22] and Symons, Irwin, Clark, and Robeck[28,30] made calculations of this nature in other publications. The author will soon be publishing a paper[26] on this type of comparison in the Wahnbach Reservoir and other European lakes that have been aerated. The stability of a stratified lake, the minimum energy input needed for destratification, is a useful concept for predicting energy requirements, inasmuch as all of the characteristics of morphology, the location of the impoundment with regard to climate, and wind influence are reflected in this parameter. To use stability as a predictive tool, stability values during summer stratification for several years prior to the mixing operation must be obtained. In this way, typical stability patterns for a particular body of water can be used to judge the energy needed for mixing, as well as the performance of any given mechanical destratification device.

Discussion of

"IMPOUNDMENT DESTRATIFICATION BY MECHANICAL PUMPING"

*Philip Cooley, James A.P. Steel, and John E. Ridley**

The authors have referred to destratification trials in the United Kingdom, where the reservoirs involved are:

1. Queen Elizabeth II reservoir (area 317 acres; depth 58.5 feet), where River Thames water is injected through "jet" type inlets at a rate of 30 mgd. Abstraction of a similar volume each day is arranged from either shoreline or central outlets. These jets were designed to promote two-dimensional circulation and prevent thermal stratification.

2. King George VI reservoir (area 350 acres; depth 52.5 feet), a standing reserve of water, for use during droughts. Thermal stratification can be prevented or eliminated, after stratification, by axial-flow pumps, mounted on rafts, capable of transferring 40 mgd from the lower layers to the upper layers.

In both these projects, modification of natural thermal patterns has provided greater volumes of high-quality outflows to waterworks, and in some respects the equipment is similar to that described by the authors. Some results of trials have been published.[29]

The authors have referred to recooling, and this is not a problem in the Thames Valley where water surfaces are well above the level of the surrounding ground. In practice, the Queen Elizabeth II reservoir can be maintained in an isothermal state by entry of less than 1 percent of its total volume each day. The inlet jets are on the floor of the reservoir and they discharge upwards at an elevation of 22°, with a water velocity of 5 fps, although a similar amount of useful work can be attributed to the wind. This prompts the writers to ask whether a simple method might be feasible in one or two of the shorter impoundments in Ohio, possibly by leading a pipe from a small weir upstream and lowering the discharge end of this pipe into the deepest part of the impoundment. The water should then circulate unattended until perhaps the stream dried up in summer, by which time little cold water would be present.

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It is of interest to compare the United Kingdom pumping system with that described by the authors (Table 5). It can be seen from Table 5 that the United Kingdom pump is much smaller in relation to the experimental environment. This is of some importance in assessing the smallest practicable pump, particularly if this work is to be carried out on large impoundments.

Table 5. COMPARISON OF PUMPING SYSTEMS

System	Pump capacity, acre-feet per day	Pump capacity, percentage of impoundment volume	Pump capacity, percentage of cold water volume
King George VI Reservoir	80	0.5	1.0
Authors			
Stewart Hollow	13	10-11	50
Caldwell	13	10-11	200
Pine	13	10-11	50
Vesuvius	13	1	10

In southern Britain, summer winds cannot destratify impoundments which are deeper than 30 feet because the epilimnion is an effective energy ''sink'' within which there is turbulent dissipation of wind-water energy interchange. As a consequence, the epilimnion-thermocline interface remains undisturbed in midsummer, although it may be slightly depressed by gale force winds.

Space does not permit discussion of eddy viscosity, eddy diffusion, and coefficients of vertical turbulence, but the United Kingdom system of discharging the colder hypolimnion water some 15 feet below the surface of the impoundment, and using for this purpose ''jet'' discharge outlets which are capable of entraining at least 10 times the discharge volume, will effectively lower a thermocline by several feet in a matter of days. The rate of depression depends on wind forces at that time, but by turbulent destruction of thermal gradients within the thermocline, the wind is capable of more useful work. In this way, the epilimnion-thermocline interface is involved in turbulent heat exchanges with water from the topmost layers.

Various systems of destratification are now accepted as a practicable means of controlling biochemical deterioration in the depths of an impoundment, but there are side effects which warrant further study. For example, increased turbulence in a water mass will hold in suspension many particles which would have normally settled to the floor of the impoundment. As a result, many more algae and particles of organic debris, with associated bacteria, then occur in the outflow, and this is of hygienic and economic significance to waterworks.

Another matter that requires attention is the rise, or fall, in the primary production of organic matter in an impoundment where partial, or complete, destratification has occurred. Experience in the United Kingdom on eutrophic waters suggests that induced turbulence, which is adequate for improving biochemical conditions in the depths, can produce biological side effects. If an impoundment is allowed to stratify, and pumps are than used to transfer water from the epilimnion-thermocline interface to the upper epilimnion, green algae disappear but certain diatoms increase in numbers. This may be caused by the transfer of nutrients specific to diatoms, but if the pump suction is then lowered into the hypolimnion this second operation results in death of the diatom bloom.

If the first period of pumping selected strata is regarded as a means of transferring ''stimulants,'' the second period might be considered as a method of transferring algal ''inhibitors.'' Although the nature and quantities of the substances involved are not known, it would be interesting to know whether the authors were planning any work along these lines.

The overall advantages of controlling thermal stratification are now recognized throughout the world, and various methods will be used at two new impoundments in the United Kingdom (20,000 to 30,000 acre-feet) and also in Hong Kong (140,000 acre-feet). It is to be hoped, therefore, that the authors will continue their excellent series of papers, which are proving to be of great value in understanding the physics, chemistry, and biology of impoundment destratification.

Discussion of
"IMPOUNDMENT DESTRATIFICATION BY MECHANICAL PUMPING"

Heinz Ambühl[*]

The problem treated by the authors is up-to-date and of the greatest importance. The results obtained with destratification are remarkable.

Similar problems are being studied in Europe. The European tests have similar aims, but they are conducted under different circumstances and conditions. Comparison with the experiments made by the authors might therefore be of interest.

Many lakes in Western and Central Europe, particularly the smaller ones, are badly eutrophized owing to urban drainage and agricultural influences, and show all the well-known symptoms such as loss of oxygen in the hypolimnion, hydrogen sulfide, and blue-green algae blooms. Since these lakes are of great importance as future drinking water reservoirs, great efforts are being made, for instance in Switzerland, to reduce sewage discharges–especially of fertilizing substances–and to find ways and means of restoring the biochemical state of such lakes.

The most striking effect of eutrophication is the hypolimnetic loss of oxygen. Many suggestions and projects have been considered over the past several decades for the aeration of lakes. However, only one process has, up to 1967, been tested on a large scale. In 1958, Müller set up an installation for the aeration of Pfäffikersee (Lake Pfäffikon near Zurich, Switzerland). The principal morphological data of this lake are as follows: greatest

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length 2.6 kilometers, greatest width 1.4 kilometers, area 3.25 square kilometers, average depth 18 meters, greatest depth 35 meters, volume 56.5 x 10^6 cubic meters, drainage basin 43.5 square kilometers (farmland, forests, villages).

The system consists of a vertical 2-meter-diameter pipe which is supported by floats and extends from a depth of 28 meters (2 meters above the bottom of the lake) almost to the surface. A diffusor at the bottom of the lake blows compressed air into the pipe; the air is supplied by an electrically powered 10-horsepower compressor which stands on the beach. The experiment lasted 5 years, and took place each year from the beginning of the stagnation period (April-May) to the beginning of the autumn circulation period (September), with a few interruptions for technical reasons. The results are summed up as follows:

1. In its natural state, the lake is strongly stratified in summer. The cold hypolimnic water was airlifted at the rate of about 6 cubic meters per second, mixed at the surface with the warm epilimnic water and stratified again, according to the temperature of the mixture, at medium depth. By this process, the lower epilimnion was warmed up, and, finally, the whole body of water as well. The temperature increase was not, however, as complete as in the experiments made by the authors and therefore did not lead to destratification. The temperature in the Pfäffikersee during the aeration process is shown at the top of Figure 15; the lower part shows the temperature in the Greifensee (Lake Greifen), a near-by lake which strongly resembles the Pfäffikersee, but which was not aerated. It is interesting to observe that the natural destratification in autumn began a month earlier than usual and reached down into the depths of the lake more quickly than in other years. This effect was observed only in the years in which the experiment took place and was not followed by any after-effect in the following year.

2. The object of the test, which was to create aerobic conditions in the hypolimnion throughout the year, was not attained. Indeed, as shown by Figure 16 the natural oxygen consumption in spring was somewhat retarded and the anaerobic phase was slightly shortened owing to the earlier start of circulation in autumn, but the artificial mixing was not extensive enough to stop it completely.

3. The artificial aeration had no immediate effect on the oxygen content, but it primarily influenced the tem-

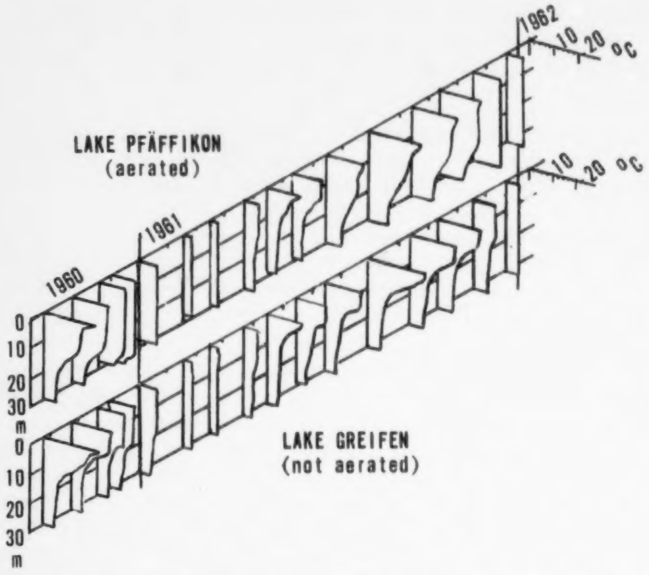

Figure 15. Temperature profiles in test and control lakes.

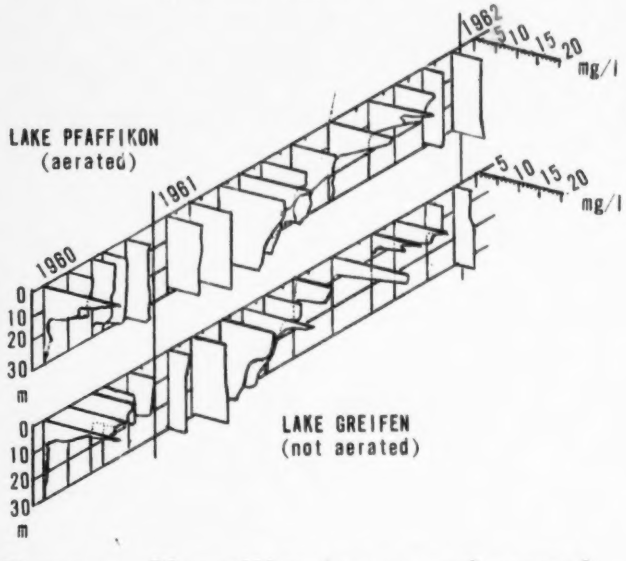

Figure 16. DO profiles in test and control lakes.

perature stratification. This result was observed throughout the lake (the airlift was installed in the middle of the lake) and confirms the observations made by the authors.

4. Before the experiment, the lake contained hydrogen sulfide during the summer stratification period. During the aeration, the hydrogen sulfide disappeared almost entirely, but was present again a few weeks later. The ammonia nitrogen content was strongly reduced and the nitrate content increased accordingly. Despite the aeration, blue-green algae *(Aphanizomenon flos-aquae)* and massive vegetation of *Peridinium* and other planktonic algae appeared.

5. The warming up of the hypolimnic water proved to be a disadvantage. This water is needed for industrial purposes and should be cold. A total warming up, as practiced in the destratification of impoundments in the United States, is not feasible in Switzerland because of legal and practical reasons. This means that the system of artificial aeration or of mechanical destratification, which is, in principle, the same, cannot be applied in Switzerland to create aerobic conditions throughout the year in strongly eutrophized lakes. On the other hand, Bernhardt[25,26] has shown that the hypolimnic water can be aerated without modification of the thermal stratification by means of a new installation. This new system could be applied with success, especially in regions where the water is needed for the preparation of drinking water and where, consequently, the prescriptions regarding temperature are very strict.

IMPOUNDMENT DESTRATIFICATION BY MECHANICAL PUMPING

Authors' Closure

The writers appreciate the interest shown by the discussers. Because this paper was the first in a series describing a 3-year research project, many of the points raised by the various discussers have been dealt with in other papers, which are included in this Compilation.[30-35]

The writers agree completely with Thackston's comments on the comparison of a high-velocity and low-velocity pump. As he suggested, only the testing of a high-discharge, low-velocity pump will determine whether or not his hypothesis is correct. Thackston wondered if diffused-air pumping might be more effective than mechanical pumping. During 1965 and 1966, mechanical- and diffused-air-pumping were compared in the same body of water.[33] On the basis of total energy input per unit of impoundment volume, the diffused-air system was about five times more effective in mixing than the mechanical pump, although both showed the same favorable improvements in water quality. The writers agree with both Thackston and Bernhardt that comparison of the actual energy input required for mixing to the theoretical minimum energy required to overturn a body of water, the stability, is a useful method of evaluating equipment performance, in spite of some limitations.[30,31,33] The writers also agree with Thackston's suggestion that maintenance of good quality throughout the summer season is preferable. This was experimentally demonstrated during the 1966 study.[33-35] Attempts at mixing when no thermal gradient exists, however, may not be quite as favorable as Thackston suggested, because the beneficial draining of the entire hypolimnion volume to the deepest portion of the lake, where the mixing device is located, would not occur, and the circulation of the water might tend to be very localized.

Bernhardt noted that 68,400 kilowatt-hours of energy input was required to reduce the stability of the Wahnbach Reservoir from its original value to a level near that of Vesuvius Lake at the end of the 1964 experiment, while an additional 36,500 kilowatt-hours was required to reduce the stability from that low level to near zero. This is in agreement with the comment made by the authors in References 30 and 33, that the rate of change of stability with time in any impoundment mixing operation declines rapidly as an isothermal condition is approached. Therefore, misleadingly poor mechanical performances will result if any equipment is not evaluated during the early phases of a destratification operation. Eight destratification experiments were analyzed by the writers,[33] and the energy input per unit of impoundment volume varied between 0.1 and 4.9 kilowatt-hours per acre-foot, averaging 2 kilowatt-hours per acre-foot. This is in good agreement with the value given by Bernhardt. The writers agree with Bernhardt's criticism of using energy input per unit of surface area as a method for estimating the size of equipment needed for destratification.

The suggestion presented by Cooley, Steel, and Ridley, to pipe warm influent water into the hypolimnion of an impoundment for release, has merit. Harnessing the incoming kinetic energy to overcome the friction losses in the pipe and the density differences of the influent and hypolimnion waters should be feasible. The only difficulty the writers envision is that an upstream weir would interfere with navigation or pleasure boating on the reservoir-river system. The writers also agree that mechanical equipment chosen for destratification should be as small as possible to be economical. During the 1965 experiment,[22] the pump used had a daily capacity of 0.45 percent of the total impoundment volume. This compares favorably with the pump used in the King George VI Reservoir. The data taken during 1965, however, indicated that the pump was somewhat undersized for this lake, as water could not be circulated rapidly enough to provide more than about 1 milligram per liter of DO in the lower one-fourth of the lake volume.

The possible biological side effects of artificial destratification mentioned by Cooley, Steel, and Ridley are of particular interest to the authors. Currently (1969) Symons and Robeck are members of the Bureau of Water Hygiene, Environmental Control Administration in Cincinnati, Ohio. One research area that will receive early attention will be the study of the biological changes that occur during artificial destratification of domestic water supply reservoirs. Because of the importance of algae (both taste and odor producing and filter clogging), bacterial and viral pathogens, and indicator organisms to the water supply industry, their behavior will be studied in stratified and artificially destratified domestic water supply reservoirs. Other water quality parameters of interest in domestic water supplies, trace organics, iron, and manganese, will also continue to be investigated. Already some work showing the temporary decline in algal populations during artificial destratification has been reported.[33,34]

Ambühl reports an early natural fall overturn in the lake that was artificially destratified. Similar results were obtained in both Boltz and Falmouth Lakes.[33] Because the general quality of the reservoir water is good as fall overturn approaches, when the overturn occurs, no serious deterioration in water quality is noted, as is often the case when heavily stratified reservoirs overturn. This is often a period of the year that severely stresses the ability of a water treatment plant to produce good quality finished water. The legal problem of maintaining cold hypolimnion water as a raw water source for domestic water supply does not occur in the United States, so overturning the entire lake volume is acceptable. In certain instances, however, cold water has some industrial uses, for example, cooling water for thermal generating plants. If maintenance of cold water were important in a given instance, some scheme such as suggested by Bernhardt[26] might be useful.

The writers are gratified in the interest that has been shown in this continuing research project, and anticipate that large, field-scale projects such as the one started in May 1968 by the U.S. Army Corps of Engineers in Allatoona Reservoir near Atlanta, Georgia will demonstrate the usefulness of this technique for maintaining good quality reservoir water.

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PAPER NO. 8

IMPOUNDMENT WATER QUALITY CHANGES CAUSED BY MIXING

J.M. Symons, W.H. Irwin, and G.G. Robeck

This paper, the second in a series of eight on artificial destratification for reservoir water quality control, discusses the data obtained during the summer of 1965. During that summer, the raft-mounted mechanical pump described in Paper No. 7 was placed in a northern Kentucky lake with a volume just under 3,000 acre-feet. The influence of artificial destratification was measured on temperature, concentrations of dissolved oxygen, manganese, sulfide, four forms of nitrogen, and phosphorus, and number and type of algae. A nearby unmixed lake of similar size and shape was used as a control for the investigation. Discussion of the manganese data that occurred during the Specialty Conference on Current Research Into the Effects of Reservoirs on Water Quality, Portland, Oregon, January, 1968 is included as an addendum.

The correct reference citation is:

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INTRODUCTION

When a flowing river is dammed and becomes an impoundment, two major changes occur and both have a marked effect on water quality. First, creating an impoundment greatly increases the time required for water to travel the distance from the headwaters to the discharge at the dam. Second, thermal or density, and, therefore, chemical, stratification may take place.

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Both the increased detention time and thermal stratification in an impoundment change the characteristics of the water discharged at a given geographical location from what they had been when the stream was free-flowing. Some of the effects of impoundments improve water quality; others deteriorate the quality. The changes resulting from impounding water have been under investigation since the 1890's,[1] and ample literature has been produced over the years.[2-12]

The success or failure of streamflow regulation using impounded water for dilution depends on the quality of the discharge water. Studying methods of controlling impoundment water quality to maximize the inherent benefits and minimize the inherent detriments is essential. Because many of the potential detriments are related to the onset of thermal stratification, artificial destratification by pumping cold water from the bottom and discharging it at the surface was suggested by Irwin, Symons, and Robeck[13] as one method of water quality control.

This paper is, then, the second in a series reporting the changes in water quality parameters in a stratified and in an artificially destratified impoundment. Through this comparison, the influence of mechanical mixing on impoundment water quality is demonstrated.

EQUIPMENT USED, IMPOUNDMENTS STUDIED , AND ANALYTICAL METHODS

The theoretical aspects of artificial destratification, and the general method used in destratification were detailed previously.[13] One change in the equipment used for destratification was increasing the size of the gasoline engine from 16 horsepower to 26 horsepower. The output of this engine at the pump operating speed was 21 horsepower. This power increase overcame the approximately 3 feet of additional head loss caused by increasing the suction line length from 30 feet to 60 feet and decreasing its diameter from 2 to 1.5 feet, while still maintaining a discharge of 13 acre-feet per day.

Bullock Pen Lake and Boltz Lake in Grant County, Kentucky, were used in this investigation. Outlines and pertinent data concerning these lakes are shown in Figure 1. Bullock Pen Lake was the control for the investigation. Analyses of weekly samples from May 14 through October 29, 1965, were made to show the changes of several common

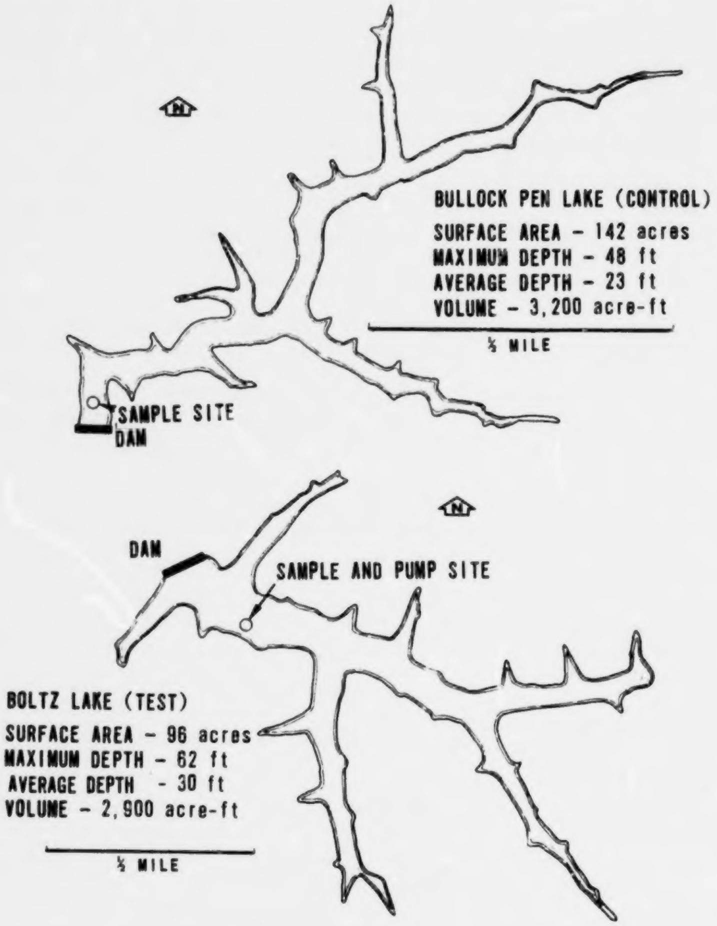
N
BULLOCK PEN LAKE (CONTROL)
SURFACE AREA - 142 acres
MAXIMUM DEPTH - 48 ft
AVERAGE DEPTH - 23 ft
VOLUME - 3,200 acre-ft
½ MILE
SAMPLE SITE
DAM
N
DAM
SAMPLE AND PUMP SITE
BOLTZ LAKE (TEST)
SURFACE AREA - 96 acres
MAXIMUM DEPTH - 62 ft
AVERAGE DEPTH - 30 ft
VOLUME - 2,900 acre-ft
½ MILE

Figure 1. Outlines of control and test lakes.

water quality parameters with time and with depth for lakes in this general area of the country. Boltz Lake, the test lake for this investigation, was sampled weekly at the pump site from May 14 to August 6, 1965. The variation of water quality measurements showed whether the control lake and the test lake were behaving in a similar fashion before the destratification operation began.

The test lake was pumped for 5 weeks between August 6 and September 10, 1965. Comparison of weekly samples from the test lake and the control lake during this period showed the influence of the mixing operation on the water quality. Finally, analysis of weekly samples from the test lake from September 10 to October 29, 1965, showed the changes in water quality that might be expected after a destratification operation had ceased.

Twenty-three different analyses were made on each of the samples taken, but only part of these data are presented herein. Because many of the methods used in this study were modifications of those in *Standard Methods*,[14] they are reported in some detail.

Temperature.—Temperatures were taken in the field with a Yellow Springs Instrument Co. resistance thermometer.

Total Suspended Solids.—Total suspended solids concentrations were determined in the laboratory by filtration through a Gooch crucible that held a glass fiber filter.

Dissolved Oxygen.—Dissolved oxygen (DO) concentrations were determined in the field using a Precision Scientific Co. Galvanic Oxygen Analyzer.

Sulfides.—Sulfide concentrations were determined in the laboratory on samples preserved in the field with zinc acetate using the colorimetric method, as given in *Standard Methods.*[14]

Manganese.—Manganese concentrations were determined in the laboratory using the persulfate method, as given in *Standard Methods.*[14]

Ammonia Nitrogen.—Ammonia nitrogen concentrations were determined in the laboratory using a Technicon AutoAnalyzer. The samples were softened by pH adjustment and filtered to remove hardness precipitate and turbidity before colorimetric reactions were made. Ammonia nitrogen was determined by reacting the samples with sodium hypochlorite and sodium phenolate.

Organic Nitrogen.—Organic nitrogen concentrations were determined in the laboratory on small samples by miniaturizing the general procedure. The procedure was: distill off the original ammonia nitrogen at a buffered pH of 7.3, digest with the acid as reported in *Standard Methods*,[14] make strongly alkaline, distill, and collect the resulting ammonia nitrogen in a weak solution of sulfuric acid. After collection, the ammonia nitrogen was determined by the procedure noted under the ammonia nitrogen section.

Total Hydrolyzable Phosphorus.–Total hydrolyzable phosphorus concentrations were determined in the laboratory by adding the hydrolyzing reagents reported in *Standard Methods*[14] to the sample and autoclaving for 30 minutes at 15 psi. The concentration of phosphorus was then determined using the stannous chloride procedure given in *Standard Methods*[14] as adapted for the Technicon AutoAnalyzer. This procedure measures all the orthophosphate, all the polyphosphate, and some of the organic phosphate. These data were then arithmetically converted to equivalent phosphorus.

RESULTS

The water quality characteristics investigated are grouped in three categories: (1) physical characteristics-temperature and total suspended solids concentration; (2) dissolved oxygen and related parameters-DO, manganese, and sulfide concentrations; and (3) nutrients and related biological populations-nitrogen and phosphorus concentrations, total algal population, and blue-green algal population. In each of the figures presented, a visual line of best fit is drawn through the data. By this technique, the actual differences or similarities between the behavior of the two lakes are more readily apparent. The control lake was only 48 feet deep, and so the 45-foot data represent the bottom of the control lake and 17 feet from the bottom in the test lake. This difference apparently did not influence the data to any great extent.

Physical Characteristics.–The temperature data shown in Figure 2 demonstrate that the upper part of the test and control lakes behaved in a similar fashion, and that mechanical pumping had little influence on the temperature pattern in these layers. At the 30- and 45-foot depths, however, the pumping operation had a marked warming influence. After pumping stopped on September 10, 1965, the test lake cooled as a unit and thermal restratification was not evident.

The total suspended solids concentration data in Figure 3 indicate that the concentration of this parameter was generally higher in the test lake than in the control, possibly because of wind action or excessive runoff; however, this increase was not aggravated by the pumping operation. Before the experiment began, the possibility was considered of increasing the surface turbidity in an

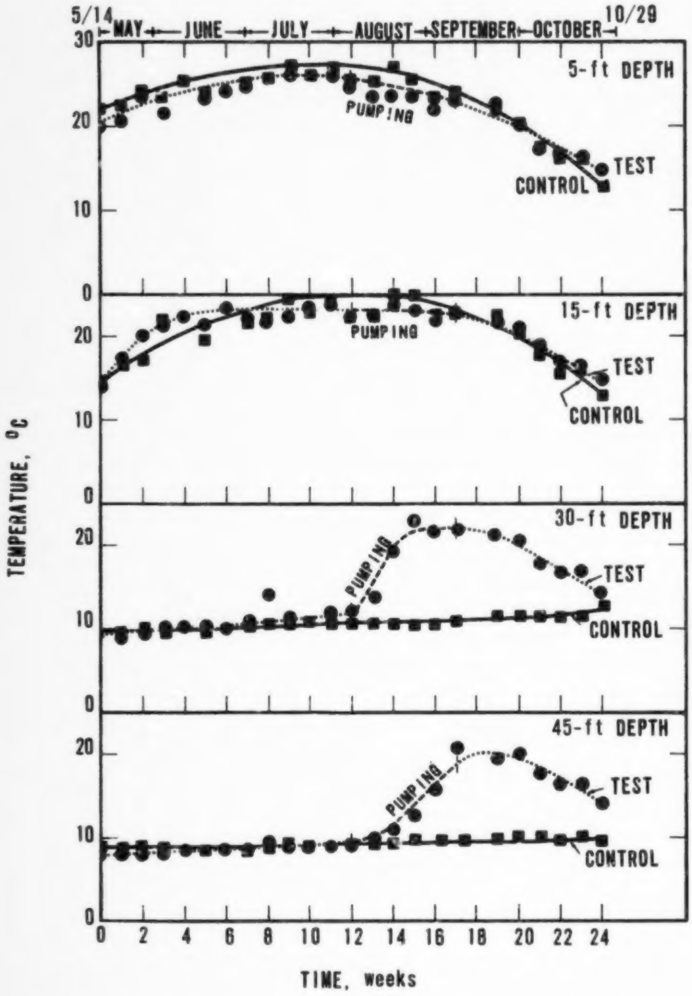

Figure 2. Changes in temperature with time.

artificially destratified impoundment by bringing solids from the bottom and discharging them at the surface. Although turbidity should have been determined, clarity was estimated from total suspended solids data (Figure 3), and seemed not to be seriously impaired.

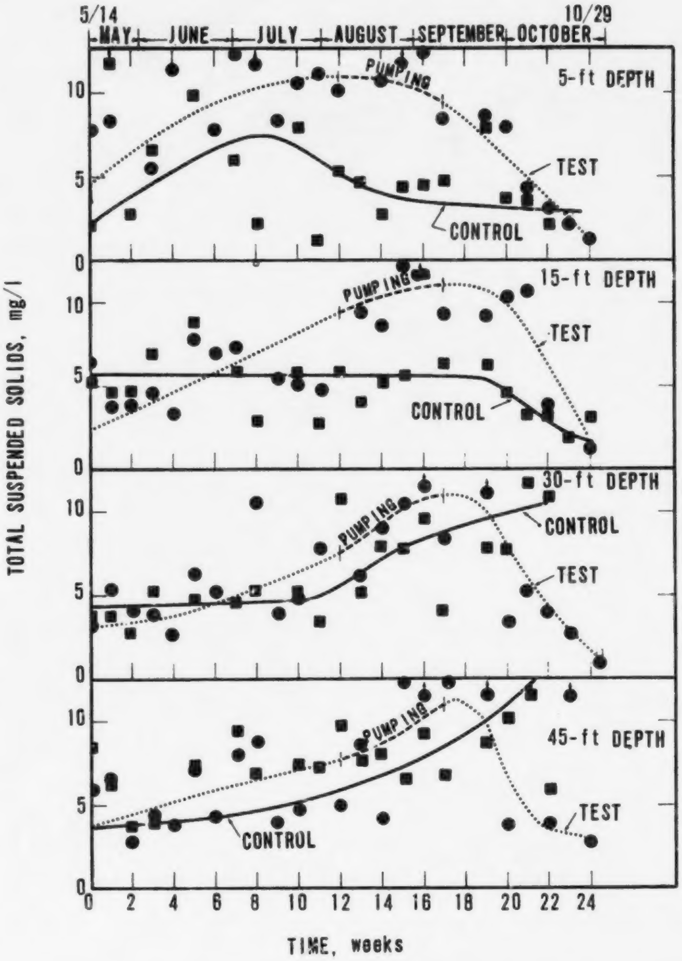

Figure 3. Changes in total suspended solids concentration with time.

Dissolved Oxygen and Related Parameters.—The DO concentration changes with depth and time (Figure 4) demonstrate that pumping had little influence in the upper zones of the lake, but had a marked influence at the 30-foot depth. Pumping had less influence on the DO concentration at the 45-foot depth, compared to the 30-foot depth, probably because of the relatively slow rate of

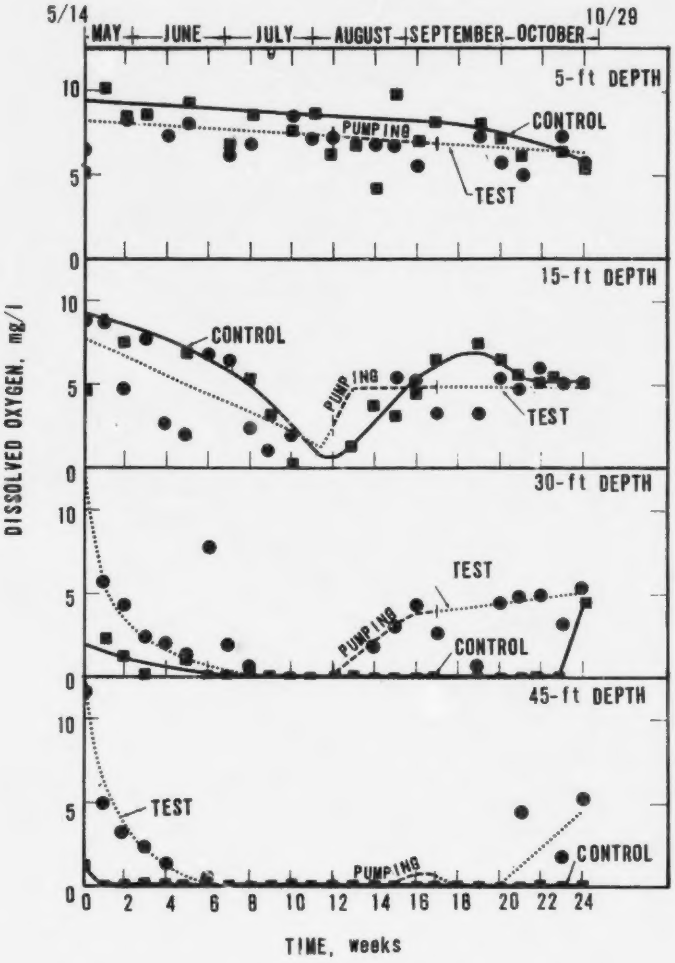

Figure 4. Changes in DO concentration with time.

artificial destratification. Because the pumping operation required 38 days to complete, the oxygen demand in the water at the 45-foot depth was almost equal to the oxygen supply rate at that depth.

Finally, the increase in DO concentration that occurred about the eleventh week at the 15-foot depth cannot be attributed to the pumping, because the control lake showed an increase at the same time. These increases were caused by a warming of the water at 15 feet that would provide an opportunity for the penetration of wind-mixing to that level.

The results of providing DO at the 30- and 45-foot depths in the test lake are shown in Figures 5 and 6. The oxidation and precipitation of manganese is shown in Figure 5; the elimination of sulfides at the 45-foot depth is shown in Figure 6. Little or no manganese occurred at the 5- or 15-foot depths in either lake, although a slight increase in concentration occurred during the fourth week of mixing; therefore, these data were not presented in Figure 5. Similarly, sulfides were detected only at the 45-foot depth in either lake; therefore, the data from the shallower depths are not presented in Figure 6.

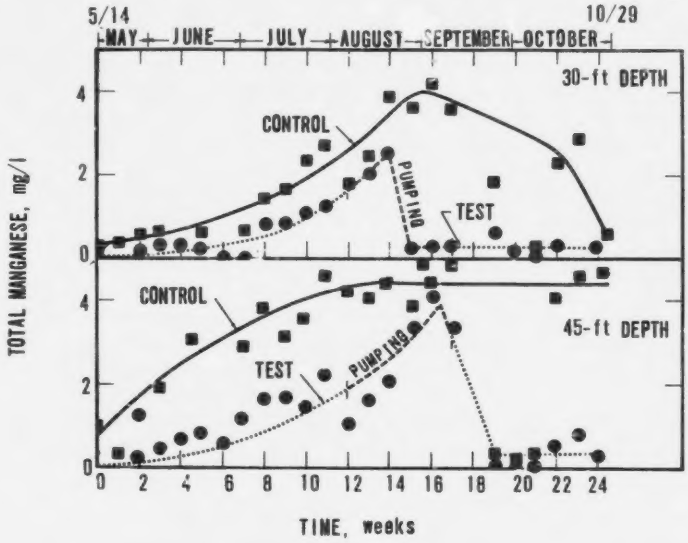

Figure 5. Changes in total manganese concentration with time.

Nutrients and Related Biological Forms.—Although artificial destratification was expected to improve the quality of impounded waters,[15,16] before this investigation was carried out, the possibility of bringing additional nutrients to the surface by mixing and aggravating the

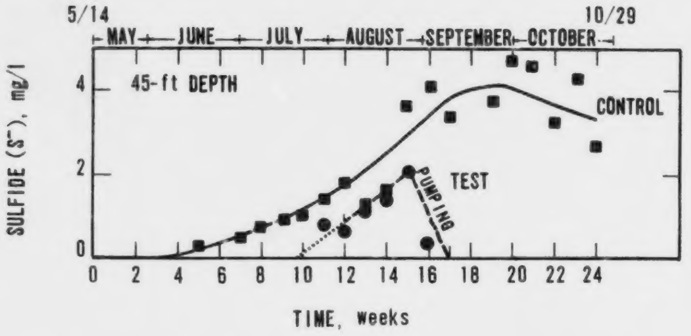

Figure 6. Changes in sulfide concentration with time.

growth of algae, was considered. The data in Figures 7 through 11, however, demonstrate that this did not occur.

The nitrogen data (Figures 7, 8, and 9) show that mechanical pumping did not significantly change the concentrations of either ammonia nitrogen, organic nitrogen, or summation nitrogen (the arithmetic sum of ammonia, nitrite, nitrate, and organic nitrogen concentrations) in the upper zones of the test lake. The only significant influence that pumping had on the nitrogen concentration data occurred at the 45-foot depth, where providing some DO helped control the increase in ammonia nitrogen concentration that would have occurred because of anaerobic decomposition had DO not been present. This same effect is shown in the summation nitrogen concentration data at the 45-foot depth (Figure 9). The absence of nitrite nitrogen and nitrate nitrogen in most of the samples from either the test or the control lake indicated that nitrification was not a strong reaction in these bodies of water.

The trend of the total hydrolyzable phosphorus concentrations (Figure 10) is somewhat different. For some reason not known at this time (1966), the total hydrolyzable phosphorus concentrations began to increase in the test lake at the 5- and 15-foot depths about the sixth week. This increase was not aggravated by the pumping, however, as the concentration started down during the mixing operation.

Because the nutrient content of the upper levels of the test lake was not increased by mixing, the algal count was not increased. Actually, the pumping operation reduced

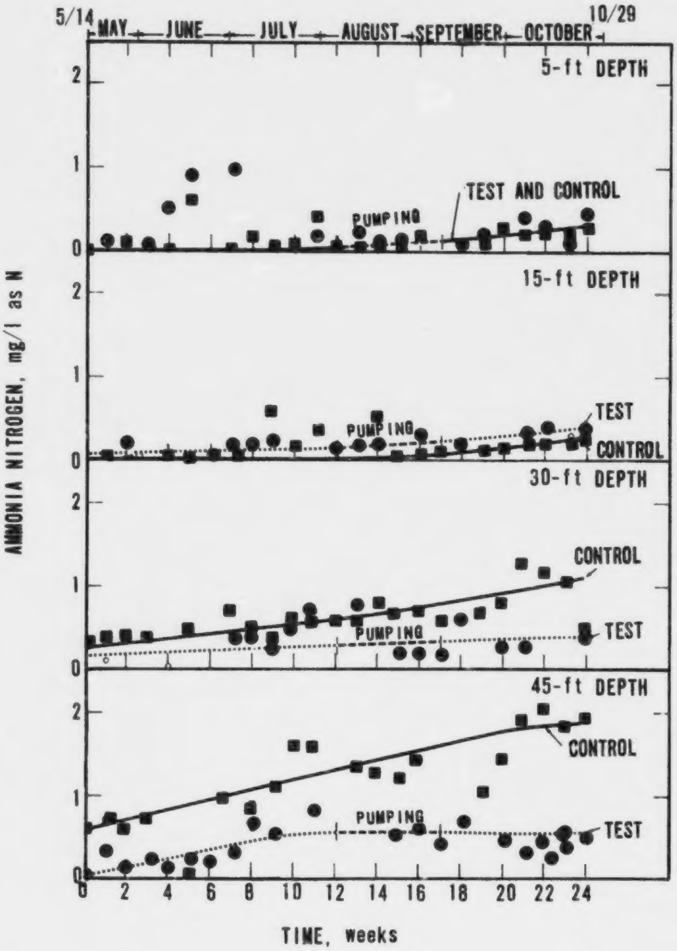

Figure 7. Changes in ammonia nitrogen concentration with time.

the number of algae in the test lake at the upper three depths (Figure 11). This reduction might have been attributed to increasing cloudiness of the water, except that the data in Figure 3 showed no marked increase in total suspended solids concentration during the pumping operation. Apparently, some environmental change occurred in the test lake water during the pumping operation that reduced the total count of algae.

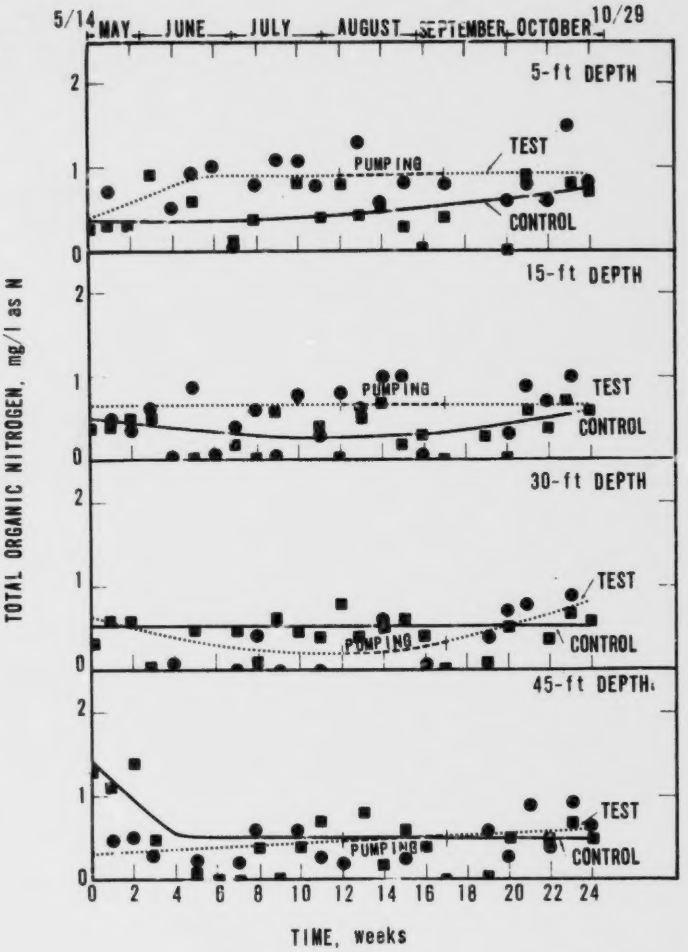

Figure 8. Changes in total organic nitrogen concentration with time.

Figure 12 shows the influence of artificial destratification on the types of algae and algal counts in the test lake. Note the format of this figure is different from that of Figures 2 through 11. The count of blue-green algae at the surface and at the 5-foot depth in both lakes starting 1 week after the pumping operation began and continuing through the remainder of the fall season is com-

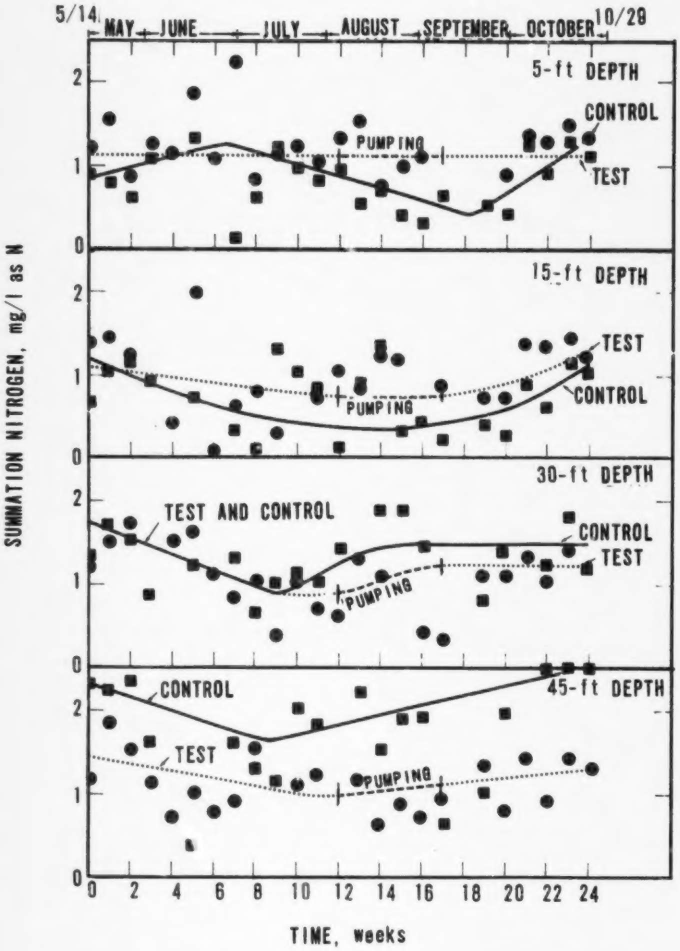

Figure 9. Changes in summation nitrogen (NH_3-N + NO_2-N + NO_3-N + Org.-N) concentration with time.

pared in the upper half of the figure, and similar data for the green algae are presented in the bottom half. These data show that the development of blue-green algae is influenced by destratification, whereas the counts of green algae did not change much in the test lake, either at the surface or at the 5-foot depth. The environmental changes that caused the decline in the blue-green-algae counts are not known at this time (1966).

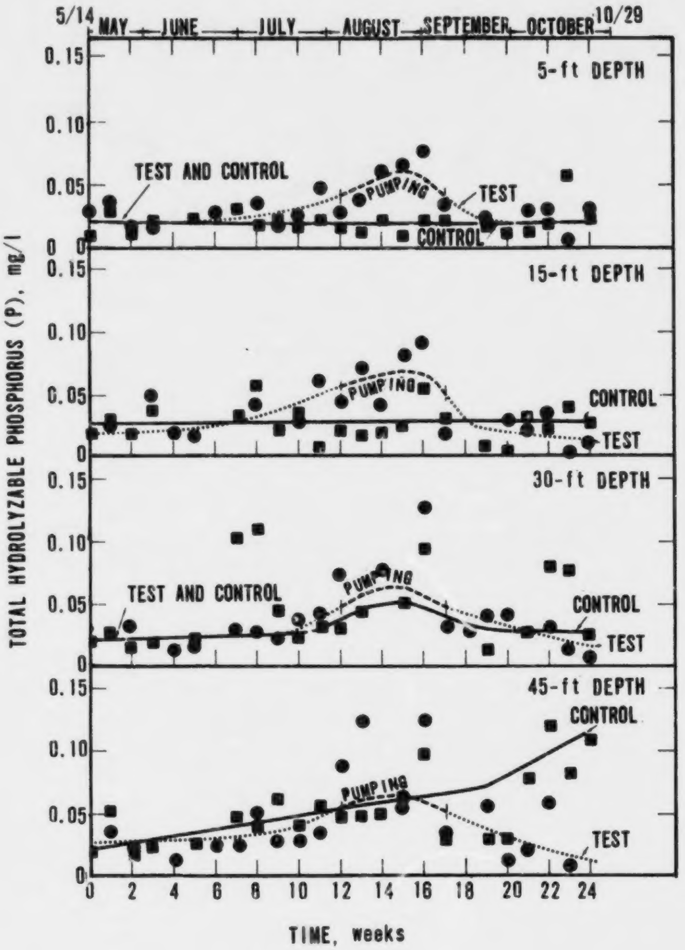

Figure 10. Changes in total hydrolyzable phosphorus concentration with time.

DISCUSSION

Impoundment Water Quality Control.—Poor quality water often develops in the lower zones of stratified impoundments. If water of this character were discharged through low-level gates in a dam, it would cause a deterioration

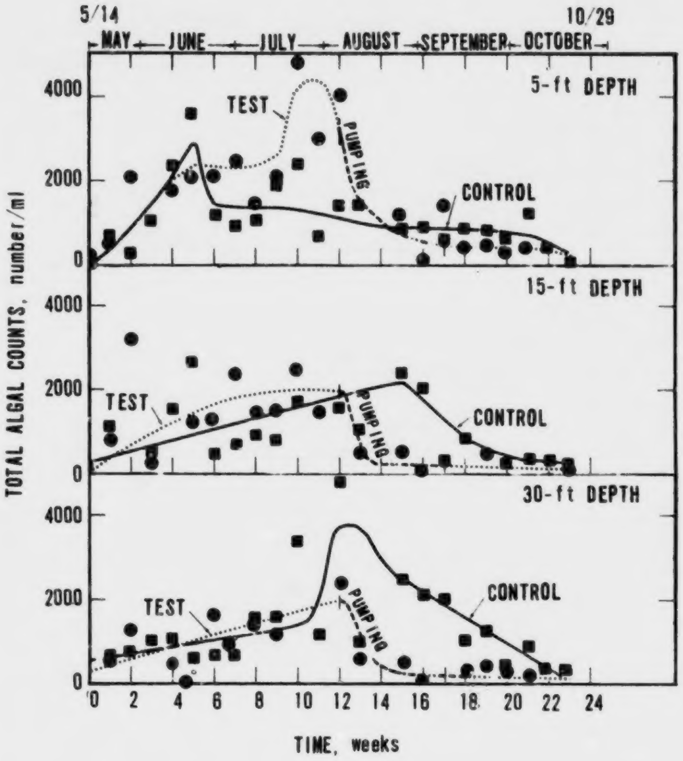

Figure 11. Changes in total algal counts with time.

rather than desirable benefits in downstream water quality. The long-range goal of this project is controlling water quality in impoundments by engineering methods. Based on previous work[13] and the literature,[15,16,17] artificial destratification of impoundments by pumping water from the bottom and discharging it at the surface seemed promising. In this study, the water quality in two lakes was measured. One lake was a control that demonstrated the behavior of some important water quality parameters in a natural stratified impoundment in the Ohio-Kentucky area. The same water quality parameters were measured in the test lake, which was unmixed during May, June, and July. The measurements showed that the test lake behaved, in general, like the control before the artificial destratification operation. During mixing, the changes in these

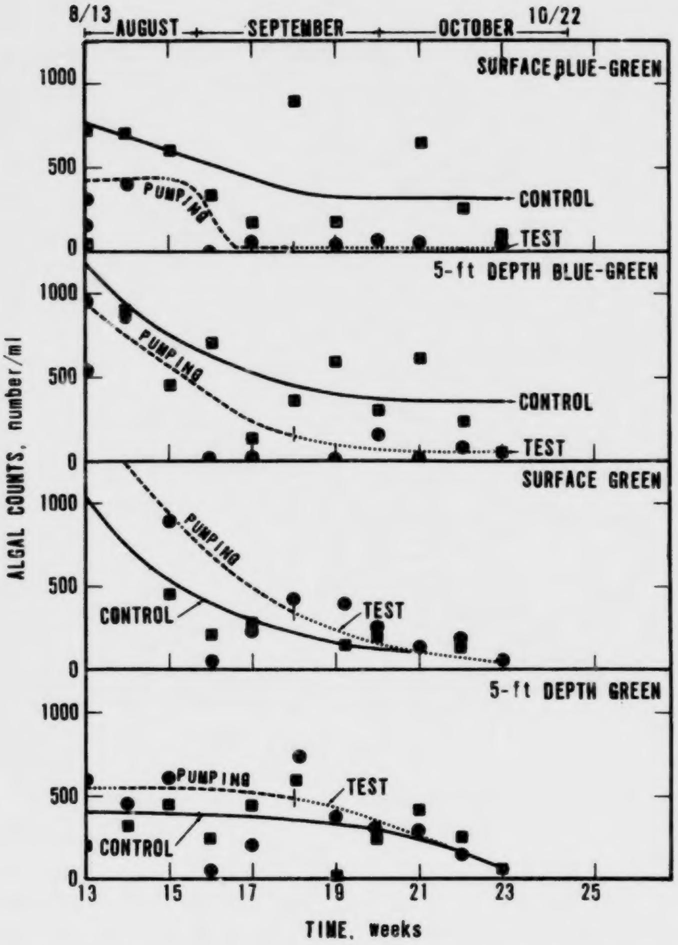

Figure 12. Changes in blue-green algal population (top) and green algal population (bottom) with time.

parameters showed the influence of mixing on the water quality.

Figures 2, 4, 5, and 6 showed the temperature and the DO concentration were increased in the lower layers of the test lake, while the manganese and sulfide concentrations were reduced to near zero. More than dilution was occurring as the total pounds of both manganese and sulfide decreased in the lake during mixing. Clarity, however, was estimated not to be impaired (Figure 3). Also, the nitrogen and phosphorus contents of the test lake surface waters were not increased, and the ammonia nitrogen concentration in the bottom layers of the test lake was reduced below that in the control by providing DO at that level.

The total algal count in the test lake was reduced by the pumping operation. The reason for this reduction is not presently known (1966), but possibly a rapidly changing environment in these surface waters upset the naturally predominating algae that had developed in the test lake before mixing began. This effect was somewhat preferential on algal type, because the decline of the blue-green-algae density in the test lake was more noticeable than the decline in the green-algae density (Figure 12).

Comparison of 1964 and 1965 Pump Performance.—The major purpose herein, as contrasted to previous work,[13] is to demonstrate the effect of artificial destratification on water quality, rather than once again describing the pumping operation itself. Because the impoundment destratified during 1965 was approximately twice the volume of the largest one pumped in 1964, some comparisons of pump performance are useful. Table 1 summarizes the data from these two operations. This table reveals one important similarity between the two operations and one important difference. The similarity may be found in line 15. In both of these operations only about one-half of the original cold water actually had to pass through the pump to mix the entire impoundment. This further reinforces the previous conclusion that currents created in an impoundment during a pumping operation are beneficial in the mixing operation.

The major difference between the 1964 and 1965 investigations is related to the total input energy per unit of impoundment volume required in these two mixing operations (line 12). In 1965, approximately three and one-half times the energy per unit volume was required to circulate the impoundment. One-half of this increase was caused by the greater volume of cold water that existed in Boltz Lake at the start of the operation, as opposed to that in Vesuvius Lake (line 13). The remainder of the increase in energy per unit volume was caused by the increased head losses in the pumping equipment. Any value given for total input

Table 1. COMPARISON OF PUMP PERFORMANCE DATA BETWEEN 1964 AND 1965 INVESTIGATIONS

Line number	Item	Vesuvius Lake	Boltz Lake
1	Year	1964	1965
2	Surface, in acres	100	96
3	Maximum depth, in feet	30	62
4	Estimated average depth, in feet	13	30
5	Volume, in acre-feet	1,260	2,900
6	Date started	9/3	8/6
7	Date finished	9/17	9/10
8	Pump capacity, in acre-feet per day	13	13
9	Input power, in horsepower	12	21
10	Time pumped, in hours	208	912
11	Input energy, in horsepower-hours	2,496	19,152
12	Input energy per lake volume, in horsepower-hours per acre-feet	1.97[a]	6.6[a]
13	Estimated volume, <20°C water,[b] in acre-feet	240 (19%)	979 (41%)
14	Volume passed through pump, in acre-feet	112	494
15	Cold water pumped, in percentage	46.8	50.5
16	Lake volume pumped, in percentage	11.2	20.6

[a] This value depends, in any given study, on the quantity of cold water in the impoundment at the start of the operation (line 13) and the performance of the equipment.

[b] Estimated volume of <20°C water in lake at beginning of pumping operation.

energy per unit impoundment volume must be viewed with caution because it is influenced both by the percentage of the impoundment volume that must be pumped to circulate the entire volume and the hydraulic performance of the pumping equipment.[13]

Minimum Energy Requirements, Efficiency, and Costs.– One method of calculating the minimum energy required to destratify an impoundment is by determining its stability.[16] Stability is defined as the work or energy needed to lift the weight of an entire body of water the vertical distance between the centers of gravity of the body of water in a stratified and in an isothermal condition. The stability of Boltz Lake was 44.4 kilowatt-hours before the pumping operation began. The stability of the control was 42.0 kilowatt-hours at this same time. During the 5 weeks of mixing, the stability of the control declined 13.0

kilowatt-hours because of deeper penetration of warm water. Over the same period, stability in the test lake was lowered 32.6 kilowatt-hours, an increase over the control of 19.6 kilowatt-hours. This reduction in stability required 14,300 kilowatt-hours input energy. The calculated ''destratification efficiency'' (DE), therefore, was 0.14 percent. Despite the low DE, the mixing operation significantly improved the water quality of the test lake (see Figures 2 through 12).

G. E. Koberg and M. E. Ford[16] destratified a 2,500-acre-foot impoundment with a total energy input of 3,400 horsepower-hours (2,540 kilowatt-hours) [1.35 horsepower-hours per acre-foot, as compared to 7 horsepower-hours per acre-foot for Boltz Lake, line 12, Table 1]. Their reservoir's original stability was 43 x 10^6 foot-pounds (16.2 kilowatt-hours), so that 0.6 percent of the compressor input energy was transferred to the water. Further, they mixed in the spring when stability is naturally increasing with time. Comparison of volumes and energy inputs may be misleading if the degree of stratification to be broken and the season of the year are not also considered.

If Boltz Lake could have been destratified by adding 31.4 kilowatt-hours (44.4 - 13.0 kilowatt-hours) of energy, and electrical energy could be obtained for $0.01 per kilowatt-hour near a hydroelectric installation, the operation cost of this destratification would have been $0.31. Scaling this cost to a 1,000,000-acre-foot impoundment (about 400 times larger) and assuming (1) an equal intensity of stratification and (2) mixing at the same season, yields an operating cost per mixing of about $125. This minimum cost is much more attractive than the $56,500 operating cost obtained from scaling the input energy cost of the Boltz Lake operation up to a 1,000,000-acre-foot impoundment.

FUTURE PLANS

Future research efforts in this area will be directed toward destratification with an energy input that approaches the minimum. One trial will be to start mixing in the spring. This is attractive because stability is low so that energy requirements are low, and reaeration is not necessary. In a springtime operation, however, a great quantity of cold water must be moved to reduce the small amount of stability that is present, and stability is naturally increasing with time. Also, an air diffusion system will be tried to determine whether the efficiencies obtained by Koberg and Ford[16] can be reached and improved. These studies will be made in 1966.[18]

SUMMARY AND CONCLUSIONS

In summary, this study has shown that artificial destratification is an effective engineering method of impoundment water quality improvement. Theoretical calculations indicate that minimum energy requirements are fairly low, but that continued research is needed to design mechanical equipment that will cause destratification with an energy input approaching the theoretical minimum, while maintaining good quality water.

The following conclusions may be drawn:

1. Artificial destratification of impoundments by mechanical pumping is an effective means of improving water quality.

2. In this investigation the control (Bullock Pen Lake) behaved in the expected manner, based on the literature.

3. In general, the test lake (Boltz Lake) and the control behaved similarly for the 3 months before the pumping began.

4. Artificial destratification warmed the lower layers of the test lake, but did not raise the total suspended solids concentration in the surface.

5. The pumping operation added DO to the lower layers of the test lake and precipitated manganese and oxidized sulfide in this zone.

6. Mixing did not change the nitrogen or the total hydrolyzable phosphorus concentrations in the surface of the test lake.

7. Mixing did not cause algal blooms in the test lake. Actually, the algal count was reduced during pumping in the test impoundment.

8. Artificial destratification reduced the counts of the blue-green algae more than that of green algae in the test lake.

9. The energy input per unit of impoundment volume was higher in this study than in the 1964 operations[13] because of a higher percentage of cold water at the start of the test and pumping equipment with lower performance.

10. Future work concerns determination of a more efficient method of artificial destratification.

ACKNOWLEDGMENTS

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ADDENDUM

Although this paper was not discussed when it was first published, discussion of the manganese data contained herein occurred during the Specialty Conference on Current Research Into the Effects of Reservoirs on Water Quality, January 1968. The topic was explored during the floor discussion that followed a paper by James J. Morgan on ''Equilibrium and Kinetic Aspects of Inorganic Chemistry Quality in Reservoirs'' and a prepared discussion of Morgan's paper by G. Fred Lee and Joseph J. Delfino. The portion of the floor discussion reproduced below appeared on pages 69-72 in the Proceedings of the Conference published as Technical Report No. 17 by the Department of Environmental and Water Resources Engineering, Vanderbilt University, Nashville, Tennessee.

Dr. James M. Symons, USPHS: I never miss an opportunity to have a specialist try to interpret my data for me. In some of the work we did on mixing, disappearance of manganese was faster than these gentlemen [Dr. James J. Morgan of the California Institute of Technology and Dr. G. Fred Lee of the University of Wisconsin in their papers presented at the Conference] have indicated. Have they looked at our data and attempted to interpret it?

Dr. G. Fred Lee, University of Wisconsin: At the time of the conference, we had not looked very closely at the data referred to by Dr. Symons in his question. In the interim period, however, we have studied the results presented by Symons, Irwin, and Robeck in the Journal of the Sanitary Engineering Division, ASCE, Paper 5205, April (1967) [this paper] and have the following comments.

The manganese data referred to by Symons are given in Figure 5. An important consideration is the length of time required for pumping-induced mixing to affect a particular depth in their test lake.

Referring to the Symons et al. (1967) data, in the third week after pumping was initiated, at 30 feet, manganese decreased in the test lake from about 2.5 milligrams per liter to about 0.2 to 0.3 milligram per liter as approximated from their Figure 5. Later in the text, Symons et al. summarized the results by saying that ''manganese concentrations were reduced to zero,'' although Figure 5 shows that there was at least 0.1 milligram per liter remaining at both the 30-foot and 45-foot depths. [This statement was corrected for this Compilation.] While these concentrations are admittedly low compared with the original 2 to 4 milligrams per liter, they are still significant in view of the USPHS accepted standards for manganese, recently placed at 0.05 milligram per liter.

In the fourth week after pumping, at the 45-foot depth, manganese was still high at about 4.0 milligrams per liter. The next point given in the Figure was for 7 weeks after pumping, with manganese showing a decrease to about 0.1 milligram per liter. The removal of manganese from the 45-foot water level appeared to be immediate from the Figure 5 plot, but there were no data for a 3-week interval and if points were available, the change probably would not have appeared to be so immediate. [The September 9, 1965, datum, 17th week or 5 weeks after pumping began, for the manganese concentration at the 45-ft depth, 3.22 milligrams per liter, was accidentally omitted from the paper Lee studied, so he could not have known that the ''TEST'' line was accurately drawn. This has been corrected.]

Symons et al. (1967) stated that little or no manganese occurred at the 5-foot or 15-foot depth in either the control or test lake and thus no data were given. But if the data at 5 and 15 feet were inspected closely they should reveal some small increases in manganese at the same time the 30-foot manganese began to decrease. This would be due to the effect of mixing and subsequent dispersion of manganese throughout the well-mixed epilimnion.

The study at Cox Hollow Lake, Wisconsin, indicated that when mixing caused manganese to decrease at intermediate depths, there was a temporary manganese increase in the upper waters. Manganese decreased in the upper waters only after the entire lake had been turned over by the compressed air system, and this manganese decrease occurred slowly over a period of several weeks. The decrease in manganese was not necessarily only the result of oxidation of Mn(II) to manganese oxide but also involved sorption of Mn(II) onto particulate iron oxides and also incorporation of Mn(II) into particulate matter matrices composed chiefly of vegetative debris.

The differences in the rates of removal of manganese between the Symons et al. (1967) data and the manganese results given in our discussion could be due to different mixing and dispersion patterns in the two lakes in addition to the intermediate role played by the particulate organic matter in Cox Hollow Lake in Wisconsin.

Dr. Symons: I appreciate the time Dr. Lee has taken to review our data. To help further clarify the discussion, Table 1 is presented. The 30-foot and 45-foot concentration data in this table were used to plot the Figure 5 to which Dr. Lee refers. The samples were unfiltered before analysis and were analyzed by the persulfate method for total manganese.

Table 1. SUMMARY OF TOTAL MANGANESE DATA DURING 1965 MIXING OF BOLTZ LAKE

Date	Days	Total Mn conc, mg/l 5 ft	15 ft	30 ft	45 ft	Mn in lake, lb	Avg conc in lake, mg/l	Remarks
7/30	0	0.02	0.24	1.33	2.25	5690	0.72	
8/6	7	0.12	0.01	1.48	1.15	5060	0.64	Mixing started
8/13	14	0.05	0.20	2.17	1.68	7510	0.95	
8/19	20	0.57	0.05	2.53	2.11	9690	1.26	
8/27	28	0.01	0.01	0.32	3.40	3310	0.42	
9/3	35	0.27	0.27	0.32	4.04	4940	0.62	Lake aerobic
9/10	42	0.17	0.19	0.34	3.22	3990	0.50	Mixing stopped
9/24	56	0.01	0.01	0.69	0.10	1950	0.25	
10/1	63	0.01	0.00	0.14	0.04	420	0.05	
10/8	70	0.01	0.01	0.12	0.10	429	0.05	

If the 5-foot datum on 8/19 is ignored, the increase in manganese concentration in the upper waters mentioned by Dr. Lee seems to have occurred. (See the 5-foot and 15-foot data on 9/3.) The reduction in concentration in the lower layers of the lake was not caused by dilution alone, however, as shown by the reduction in the total number of pounds of manganese in the lake (Table 1).

In an attempt to compare the behavior of total manganese in Boltz Lake to the total manganese behavior in Cox Hollow Lake, Table 2 is reproduced from the paper ''Limnological Changes Resulting from Artificial Destratification and Aeration of an Impoundment,'' by T. L. Wirth and R. C. Dunst, Research Report No. 22 of the Wisconsin Conservation Department, March 1967. As Dr. Lee mentioned, an increase in manganese concentration occurred at the 3-, 6-, and 9-foot depths after mixing started, as shown by the 7/14 data. As in our experiment, however, more than simple dilution was taking place, as the total pounds of manganese in the lake did decrease after 7/14.

Table 2. SUMMARY OF TOTAL MANGANESE DATA
DURING 1966 MIXING OF COX HOLLOW LAKE

		Total Mn conc, mg/l										
Date	Days	3 ft	6 ft	9 ft	12 ft	15 ft	18 ft	21 ft	24 ft	Mn in lake, lb	Avg conc in lake, mg/l	Remarks
6/29	0	0.12	0.12	0.19	0.69	1.39	0.84	1.42	1.53	1390	0.43	Mixing started 7/1
7/14	15	0.26	0.37	0.50	0.67	0.94	1.00	1.84	2.30	1840	0.57	
7/28	29	0.26	0.24	0.30	0.32	0.41	0.37	0.27	0.50	970	0.30	Lake aerobic 7/18
8/18	49	0.14	0.14	0.19	0.19	0.19	0.20	0.20	0.26	535	0.17	
9/22	84	0.09	0.09	0.10	0.11	0.10	0.09	0.10	0.11	308	0.10	

In an effort to compare the rate of disappearance of total manganese from both lakes, the average concentration of total manganese (the total pounds divided by the total lake volume) was calculated for various dates. These data are also presented in Tables 1 and 2. In Boltz Lake, the rate of decline of the average manganese concentration from the time the lake became aerobic, day 35, to day 63, was fairly constant and averaged 0.022 milligram per liter per day. In Cox Hollow Lake, from day 29 (4 days before the lake became aerobic) to day 84, the rate of decline of the average Mn concentration was also fairly constant, and averaged 0.004 milligram per liter per day.

In summary, true disappearance of total manganese was occurring in both lakes, and the rate of disappearance was approximately five times faster in the Boltz Lake experiment than in the Cox Hollow experiment. The reason for this faster rate may well be those given by Dr. Lee in his comments.

Dr. James J. Morgan, California Institute of Technology: I think the rate of disappearance would be a very strong function of the total suspended material in the system. If you had the right kind of suspended material at a sufficiently high level, I believe the rate of disappearance would be relatively rapid. In our laboratory experiments, the rate of dissolved manganese disappearance was quite rapid with different types of solids, and at a lower concentration, the disappearance rate could be low. If you had a higher concentration of solids, all other conditions remaining the same, I think the rate of disappearance would be high. I believe that is the difficulty in trying to generalize from isolated experiments. In this particular case, I think that because there were three mechanisms that could be involved–oxidation, sorption, and precipitation–you have to determine which ones are involved and whether or not all of the suspended solids are similar. I believe all the data are ''right''; I think the systems are different.

PAPER NO. 9

MANAGEMENT AND MEASUREMENT OF DO IN IMPOUNDMENTS

(Engineering Portion)

J.M. Symons, W.H. Irwin, R.M. Clark, and G.G. Robeck

This paper, the remainder of Paper No. 4 which was started in the "Investigative Research" portion of this document, is the third in the series of eight on the use of artificial destratification for water quality control in reservoirs. Here, the theoretical basis for judgment of mechanical performance of any destratification device using "destratification efficiency" (DE) based on reservoir stability and "oxygenation capacity" (OC) is presented. The 1965 mixing data are reinterpreted with emphasis on the influence on the oxygen budget. This material was presented at the National Symposium on Quality Standards for Natural Waters held at the University of Michigan in 1966. Two discussions and a closure are also reprinted.

The correct reference citations are:

Symons, J. M., W. H. Irwin, R. M. Clark, and G. G. Robeck. "Management and Measurement of DO in Impoundments." Journal Sanitary Engineering Division, American Society of Civil Engineers, 93, SA6, 181-209. December 1967.

Thackston, E. L. Ibid. 94, SA4, 748-52. August 1968.

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Closure. Ibid. 95, SA2, 342-49. April 1969.

INTRODUCTION

The problem of thermal stratification in storage reservoirs is widespread throughout the United States. Considerable attention has been given to developing engineering methods that will prevent downstream problems from the discharge of poor quality bottom water from impoundments, particularly during peaking power operations. Several possible design features that might be incorporated into dams and outlet works to prevent the escape of poor quality water from impoundments have been extensively reviewed by Kittrell.[1] One of these techniques, artificial destratification, has been researched during the past 2 years by staff members of the Cincinnati Water Research Laboratory.

The first publication on this subject[2] discussed the theoretical aspects of artificially breaking the thermal stratification of an impoundment, and presented early experimental results. The second paper[3] dealt with a larger field trial, and extensive data were presented to demonstrate the improvement in water quality that could be effected by pumping cold water from the bottom of a stratified impoundment and discharging it at the surface, thereby creating an overturn in the body of water.

Although to date (1967) the method has been tried only on relatively small bodies of water (3,000 to 4,000 acre-feet), artificial destratification is an effective method for water quality improvement. This technique is doubly attractive because it improves the water quality in the reservoir itself as well as protecting downstream quality. This can be contrasted with some of the techniques discussed by Kittrell[1] that only prevent poor quality water from escaping an impoundment, and do not prevent the formation of poor quality water in the impoundment itself.

Research on this process will continue, and one purpose of this paper, the third in the series, is to establish a firm theoretical basis for comparing the hydraulic performance of various pieces of equipment that might be used for artificially destratifying a body of water. Secondly, although dissolved oxygen (DO) is not the only quality parameter in natural waters, it certainly is an important one. Therefore, the writers' other purpose is to examine the influence that artificial destratification has on the DO resources in an impoundment from three aspects: (1) The elimination of reduced substances; (2) the addition and redistribution of DO; and (3) when artificial destratification should be used for quality control during a summer season.

THEORETICAL CONSIDERATIONS

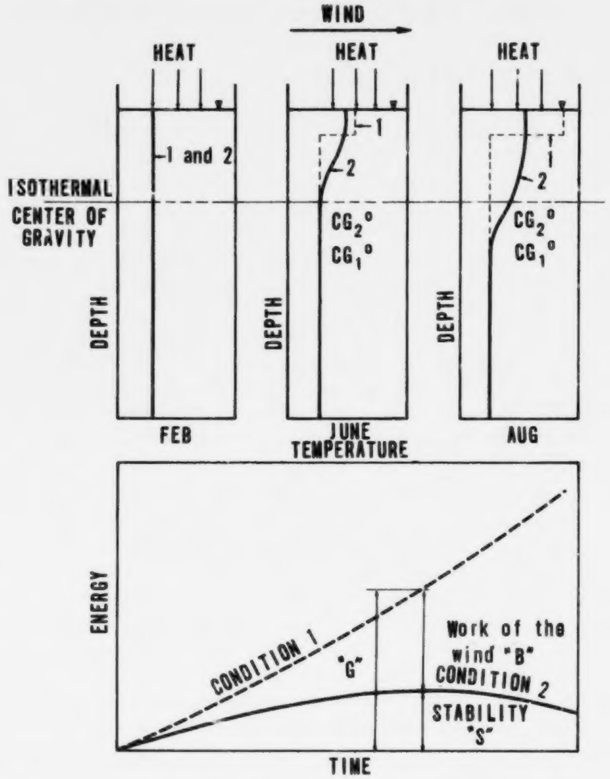
Hydraulic Performance

There are four major methods currently (1967) available for artificial destratification: mechanical pumping,[2,3] the Air-Aqua system,[4] compressed air,[5] and the Aero-Hydraulic Gun.[6] If these four systems are to be compared, their hydraulic performance must be judged on an equal basis, through the calculation of their ''destratification efficiency'' (to be defined subsequently herein). The hydraulic performance of any destratification device is evaluated by comparing the actual energy input necessary to cause overturn in a given body of water with the theoretical minimum energy required to overturn that body of water. The theoretical minimum energy required for artificial destratification is defined as the stability of the body of water.

The stability of stratified impoundments has been discussed by Koberg and Ford,[5] and by Hutchinson,[7] and can be calculated according to the following definition: the work or the energy required to lift the weight of the entire body of water the vertical distance between the center of gravity of the water mass when isothermal and the mass center of gravity when the impoundment is thermally stratified. For the purpose of this paper the center of gravity is taken as a point anywhere on a horizontal plane that passes through the true mass center of gravity.

Figure 1 schematically illustrates the stability concept. In the upper portion the temperature profile in a column of water, taken as a rectangular slice from a wedge-shaped body of water, is shown at three different months. In the first column, the water is isothermal, and the center of gravity along the vertical axis is approximately one-third the depth of the water column. The isothermal center of gravity does not change throughout the summer season. In the subsequent discussion, movement of the position of the center of gravity is considered along the vertical axis only.

As the water column is heated from the surface, two different temperature profiles, labeled 1 and 2, are shown for June and August. Profile 1 in June, dotted, would occur if there were no wind blowing across the surface of the water. In this condition, most of the incident heat would be stored in the upper layers of the water, making this water lighter relative to the remainder of the water column. This decrease in weight would lower


WIND
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CG_1°
DEPTH
DEPTH
DEPTH
FEB
JUNE
TEMPERATURE
AUG
ENERGY
CONDITION 1
"G"
Work of the
wind "B"
CONDITION 2
STABILITY
"S"
TIME

Figure 1. Schematic diagram to demonstrate stability and work of the wind.

the center of gravity of the stratified water column to the position shown by CG_1. As heating continues into August, thermal stratification becomes more intense, and the center of gravity moves down into the water column even farther. Because the actual center of gravity continues to lower throughout the heating season, the energy required to mix the water column, which is evaluated by multiplying the weight of the water column by the distance between the isothermal center of gravity and CG_1 at any given time, continually increases with time. This is shown in the lower portion of Figure 1, by the curve labeled ''condition 1.''

Wind is always present in any natural situation. The influence of wind blowing across the surface of the water column changes the temperature profile from the condition labeled 1 to the condition labeled 2. The surface of the water column does not become as warm, and the warm water extends farther down into the water column. At any given time, therefore, the center of gravity of the stratified water column does not move down to the level of CG_1, but descends only to the level shown in Figure 1 as CG_2. Because wind has the effect of preventing stratification from being as intense as it would be without wind, the actual energy necessary to mix the water column is considerably reduced because of wind action. This is shown in the lower portion of Figure 1, where the condition 2 curve is considerably below the condition 1 curve. The distance between these two curves has been defined as ''work of the wind.''[7]

In summary, at any given time, the energy that is represented by the distance, *G*, in Figure 1 is the energy that would have been required to mix the stratified water column in the absence of any wind. Since the wind has put into the water the quantity of work indicated by dimension *B*, only the remainder, the stability, *S*, must be supplied by any mechanical device, to make the water column isothermal. In any natural situation, the stability can be arithmetically calculated.[5] This theoretical minimum energy requirement for destratification, when compared with the energy input of a given mechanical device that caused destratification, defines the hydraulic performance of the mechanical equipment.

The stability curve, *S*, in Figure 1 also shows that stability reaches a maximum in an impoundment during the middle of the heating season, usually in late July, and then begins to decline. The maximum stability in a body of water occurs when the warm water has penetrated to the depth of the isothermal center of gravity along the vertical axis. As the water warms below this depth and the water body approaches an isothermally warm condition, the distance decreases between the isothermal and the stratified centers of gravity and the stability declines, reaching zero when the water body is again isothermal but warm.

Problems with the Stability Concept

While stability is a useful method of calculating the hydraulic performance of an artificial destratification device, it must be used carefully to avoid misinterpretation. Three problems are considered. The first is the effect of size and geometry on the calculation of stability. Table 1 summarizes data from three lakes near Cincinnati, Ohio, that are under study. Columns 2 through 6 list the pertinent physical data for these lakes. The stability of each lake, Column 7, was calculated assuming the same temperature profile in each lake, 20°C for the upper 5 feet of water, and remainder of the water body at 4°C. As seen in Column 7, the stability of these three lakes is different.

Table 1. COMPARISON OF STABILITY IN LAKES WITH THE SAME ASSUMED TEMPERATURE PROFILE BUT DIFFERENT SIZE AND GEOMETRY

Lake (1)	Surface, acres (2)	Greatest depth, feet (3)	Average depth, feet (4)	Vertical isothermal center of gravity, feet (5)	Volume, acre-feet (6)	Stability, kilowatt-hours (7)	$\frac{\text{Stability}}{\text{area}}$, watt-hours per acre (8)	$\frac{\text{Stability}}{\text{volume}}$, watt-hours per acre-foot (9)
Falmouth	225	42	20	14.232	4600	21.8	96.9	4.7
Bullock Pen	142	48	23	15.058	3200	15.5	109.2	4.8
Boltz	96	62	30	20.867	2900	15.0	161.3	5.2

Hutchinson[7] suggested that stability should be calculated per unit of lake surface, to eliminate differences of size and geometry. Column 8 shows, however, that this is not a reliable method of calculating a comparable parameter. At least for these three lakes, the stability per unit volume, Column 9, is a better method of obtaining a parameter that can be compared from lake to lake. Table 1 demonstrates that the stability calculated for one body of water cannot be assumed to exist in other lakes and reservoirs in the same general geographic location. To evaluate properly the hydraulic performance of a mixing operation in a certain impoundment, the stability for that particular impoundment must be calculated. Table 1 also illustrates the difficulty in using a second nearby impoundment as a control for comparison purposes in an artificial destratification study. Differences in size and geometry between the two impoundments may prevent the second from being used as an exact control.

Second, the continuing natural change that the stability of a given body of water undergoes throughout a summer season makes its use for the calculation of hydraulic performance difficult. As shown in Figure 1, as stratification begins, the stability rises from zero, reaches a maximum in late July, and declines towards zero again because of continued wind-mixing and nighttime cooling in August and September. Therefore, if the stability in a given impoundment were calculated when a mixing operation began, the natural change in stability occurring during the time of the mixing operation must be included in any proper evaluation of hydraulic performance.

If a destratification operation should take place in the spring, and should require a month to complete, the actual energy input during mixing should be compared with the calculated stability at the start of the operation, plus the natural increase in stability that would have taken place during the month. During a fall mixing operation, the natural decline in stability with time should be subtracted from the initial stability, to evaluate properly the hydraulic performance of the overturn operation.

Third, the hydraulic inefficiency of a spring destratification operation, in which cold water is pumped from the bottom to the surface, must be recognized. This is illustrated by the data in Table 2. Here, temperature profiles were assumed for four different months during the year and the stability calculated for each of the profiles, (see Column 4). The energy that must be added to the lake for mixing is low in the spring and fall and high in midsummer.

If destratification is to be carried out by bringing cold water from the bottom as a heat sink to absorb heat

Table 2. VARIATION IN PUMPING REQUIREMENTS TO DESTRATIFY BULLOCK PEN LAKE AT DIFFERENT SEASONS

Month	Assumed temperature profile		Stability, kilowatt-hours	Cold water volume required to mix lake, acre-feet	Cold water volume required to mix the lake per unit stability, acre-feet per kilowatt-hour
	Depth, feet	Temperature, °C			
(1)	(2)	(3)	(4)	(5)	(6)
April	0 to 5	20	13.8	2,560	180
	5 to 45	9			
June	0 to 15	23	32.9	1,440	43
	15 to 45	9			
August	0 to 25	25	28.2	640	23
	25 to 45	10			
September	0 to 40	25	2.5	35	15
	40 to 45	10			

from the upper layers and thereby make the lake isothermal, the volume that must be moved is the volume of cold water that is present at the start of the operation. These volumes are shown in Table 2, Column 5.

Although the volume of cold water that must be pumped in the spring is high, the work requirements are not, because the elevation head through which the water must be lifted, based on density difference,[8] is much smaller in the spring than in the summer or fall. Practically, however, mixing equipment cannot take advantage of this low elevation head in the spring and so the actual energy input at any time is directly proportional to the volume of water that must be moved (Table 2, Column 5). Therefore, although theoretical energy requirements are low in the spring, a hydraulically inefficient operation results because of practical considerations. The improvement in pumping requirements as the summer season progresses is again shown in Table 2, Column 6, where the volume required to mix the lake is calculated per unit of original stability.

The practical inefficiency of destratification in the spring could be overcome by pumping the smaller volume of warm water down into the colder layers. The theoretical energy requirements would be the same, because pumping cold water up involves moving a large volume through a small elevation head, whereas pumping warm water down would result in moving a small volume through a larger elevation head. As noted before, however, most mechanical equipment cannot take advantage of the elevation head differences; therefore, the actual input energy is directly proportional to the water volume moved. This would make pumping warm water down in the spring a hydraulically more efficient operation.

This variation in pumping requirements with changing seasons does not eliminate the usefulness of the stability concept for calculating hydraulic performance; the above should indicate, however, that a low performance value may occur during a spring destratification operation and that comparing mixing equipment during different seasons is unfair.

Another practical consideration in any mixing operation is the change in hydraulic performance as the mixing nears completion. In Figure 2 from the work of Koberg and Ford,[5] the rate of change in stability per unit of time decreases as the stability approaches zero. There are at least three reasons for this. First, because so much of the water in

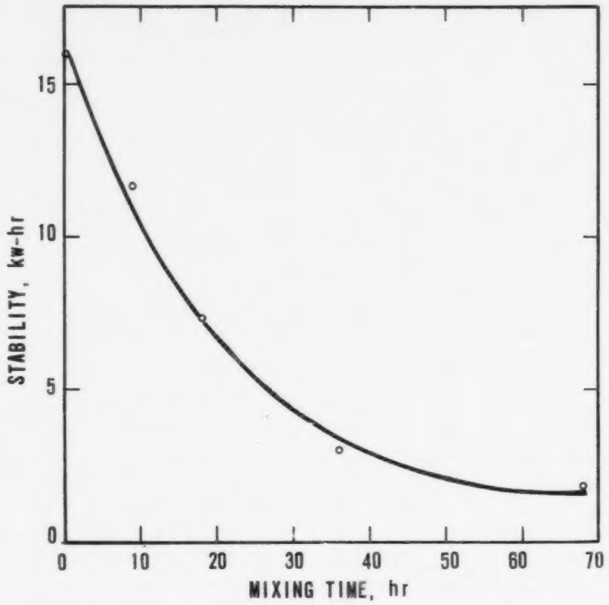

Figure 2. Decrease in stability with application of compressed air (Reference 5).

most lakes is in the upper portions, a slight temperature gradient can produce some stability. Practically, complete elimination of stability, particularly in the spring, may be impossible. Second, as mixing nears completion, the density of the body of water is nearly uniform. Under these conditions some of the water that has been raised to the surface may sink to the bottom and be repumped. This would lead to excessive energy inputs to complete the destratification.

Finally, as the epilimnion of a body of water is cooled to near its final temperature, the decrease in stability per unit volume of water pumped is smaller. This is caused by the nonlinear change in water density, which results in a nonlinear change in stability, with changing temperature.[8] In the example shown in Table 2, for the April data, the first 155 acre-feet of bottom water pumped up decreases the stability 2.3 kilowatt-hours, while the last 155 acre-feet of water mixed only eliminates 0.8 kilowatt-hour of stability. (Note, the smaller the difference between the initial and final temperatures of the epilimnion, the less pronounced this effect would be, because the incremental change of water density with temperature tends to be uniform over small temperature changes.) Keeping a continuous record of stability during a destratification operation is recommended to avoid excessive energy inputs in the late stages of the mixing.

EQUIPMENT AND METHODS

The equipment and methods used in this study have been explained in detail by Symons, *et al.*[3] and will be briefly summarized here. Artificial destratification was studied from August 6 to September 10, 1965, in Boltz Lake, a 2,900-acre-foot impoundment near Cincinnati, Ohio. Destratification was accomplished by pumping cold water from within 2 feet of the bottom and discharging it in the surface with a 13-acre-foot per day capacity, raft-mounted pump, driven by a 15.5-kilowatt gasoline engine. Analysis of temperature, and DO, sulfide, manganese, and total COD concentrations at various depths, weekly from May 14 to October 29, 1965, demonstrated the changes in stability and DO resources that occurred in this impoundment before, during, and after the pumping operation. A nearby, unmixed lake, Bullock Pen, served as the control. Physical details of both lakes are summarized in Table 1.

RESULTS

Hydraulic Performance

The approximate magnitude of the work of the wind in the control lake, Bullock Pen, during the 1965 season is shown in Figure 3. The *G* energy was approximated by assuming that all the heat incident to the lake was concentrated in the upper 5 feet of the water. The *G* energy is much higher than the stability, the lower curve, indicating that, at least for this impoundment, the wind would provide more of the total energy needed for mixing than would have to be provided by mechanical equipment.

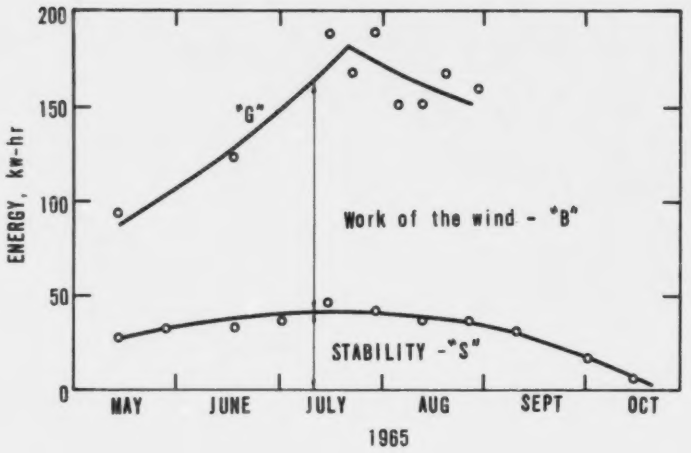

Figure 3. Estimation of the work of the wind in Bullock Pen Lake (control lake).

Figure 4 compares the stability pattern for the test and the control lakes. The control lake, Bullock Pen, shows the typical stability pattern, reaching a maximum stability of 45 kilowatt-hours in mid-July. At that time the calculated vertical distance between the isothermal and actual centers of gravity was 0.17 inch. The test lake, Boltz Lake, reaches its maximum stability at about the same time, but the calculated stability for Boltz Lake is somewhat above that for the control throughout the unmixed period because of differences in size and geometry. In spite of this slight inconsistency, Bullock Pen Lake was assumed to be a reasonable control for this operation.

Artificial destratification of Boltz Lake was started on August 6, 1965, when the pumping equipment became available. As shown in Figure 4, through the 5 weeks of pumping, the stability in Boltz Lake dropped more sharply than the natural decline of the control. After the 5 weeks of pumping, the calculated stability in the test lake was 18 kilowatt-hours lower than the control at the same time. Therefore, the destratification efficiency of this particular piece of mixing equipment, calculated according to Equation (1), was a rather low 18/14,300 = 0.13 percent.

$$\left[\left(\frac{\text{Net Change in Stability from } t_1 \text{ to } t_2}{\text{Total Energy Input from } t_1 \text{ to } t_2}\right) \times 100\right] = \begin{matrix}\text{Destratification}\\ \text{Efficiency}\end{matrix} \qquad (1)$$

in which t_1 = time of start of mixing, and t_2 = time of end of mixing.

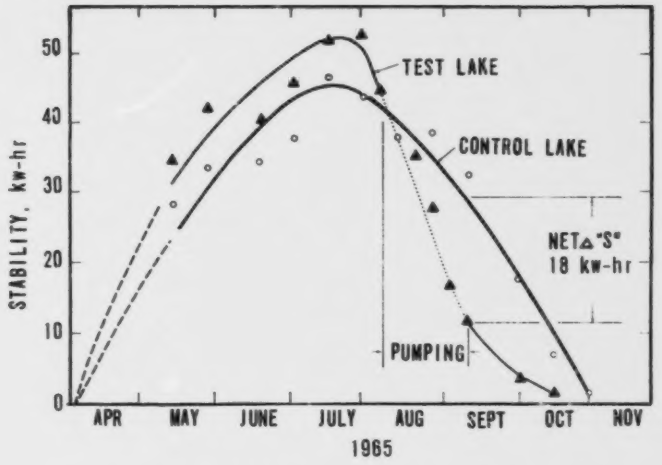

Figure 4. Stability in test and control lakes.

Elimination of Reduced Substances

In Figure 5, the total quantity of sulfide in the control lake, calculated as the equivalent oxygen demand, rises throughout the summer, reaching a peak in mid-September. After this time, the natural overturn of the lake provides DO throughout the depth, thus oxidizing and eliminating the sulfides. The quantity of sulfides in the test lake rises similarly to that in the control prior to the artificial destratification operation. As pumping provided DO throughout the depth of the test lake, the sulfides were oxidized and their quantity lowered, as shown in the dotted portion of the test lake curve. Assuming that the sulfide level of the test lake would have risen to approximately that of the control lake on September 10, the net change in the sulfide content caused by pumping was 13,000 pounds, as the oxygen equivalent. The oxygen equivalent of the sulfide was calculated as twice its concentration, since two moles of oxygen are required to oxidize one mole of sulfide.

A similar, but smaller, change is shown in the manganese oxygen demand in Figure 6. For this analysis, all of the manganese measured in both lakes was taken to be soluble, divalent manganese, and the oxygen equivalent of the manganese was calculated as 0.29 of its concentration,

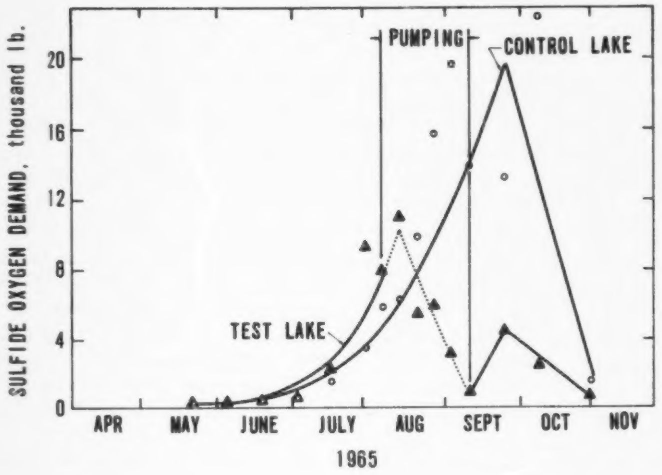

Figure 5. Total sulfide as oxygen equivalent in test and control lakes.

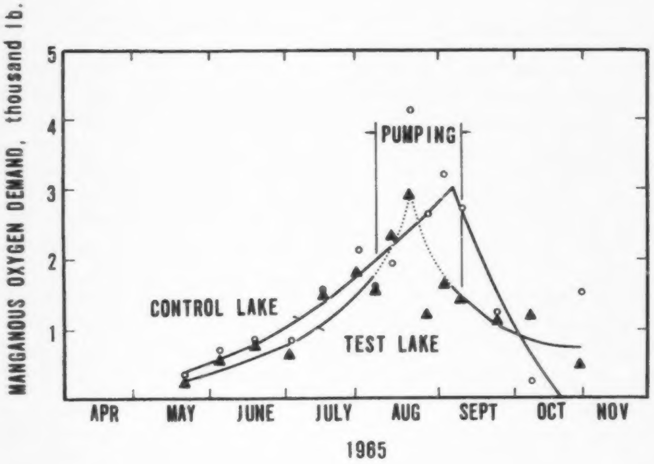

Figure 6. Total manganese as oxygen equivalent in test and control lakes.

because one-half mole of oxygen is required to oxidize one mole of manganous manganese. The net change in the manganese oxygen equivalent between the control and test lakes at the end of the pumping operation, September 10, was 1,300 pounds of oxygen equivalent. As with the sulfides, the oxidation of the manganese in the test lake was accomplished by the destratification providing DO to the lower level of the lake.

The total COD content of the test and control lake was also calculated for each week during the study period. These data were more scattered than those shown in Figures 5 and 6, however, and no significant reduction in the total COD content in the test lake could be noted during the pumping period. This is probably because the degradable portion of the total COD was so small that even though the addition of DO to the lake enhanced the biodegradation of this portion of the organic material, the total COD measurement probably was too insensitive to detect any biodegradation.

In summary, the artificial destratification operations on Boltz Lake, from August 6 through September 10, 1965, supplied a total of 14,300 pounds of DO, which satisfied an equivalent quantity of inorganic oxygen demand.

Addition and Redistribution of DO

The decline in the total quantity of DO in both lakes through the early part of the summer season because of continuing oxygen demand in the water is shown in Figure 7. When pumping began in the test lake, the quantity of DO rose, but at the same time, because of continued wind-mixing, additional DO was appearing in the control lake. Based on these data, although 14,300 pounds of DO were supplied to the test lake (Figures 5 and 6) the pumping capacity of the equipment was not sufficient to add the desired quantity of DO above the inorganic oxygen demand. The oxygen demand caused by the natural fall overturn in late September is also shown in Figure 7. This occurred even in the test lake, which restratified slightly after the pumping operation stopped.

Although the desired quantities of DO beyond the inorganic oxygen demand were not added to Boltz Lake, Figure 8 shows the redistribution of the existing DO resources caused by the pumping operation. Here, DO, absent below 20 feet before pumping began, was present, although in low concentrations, at all depths after pumping. Because the goal of any destratification operation is to provide

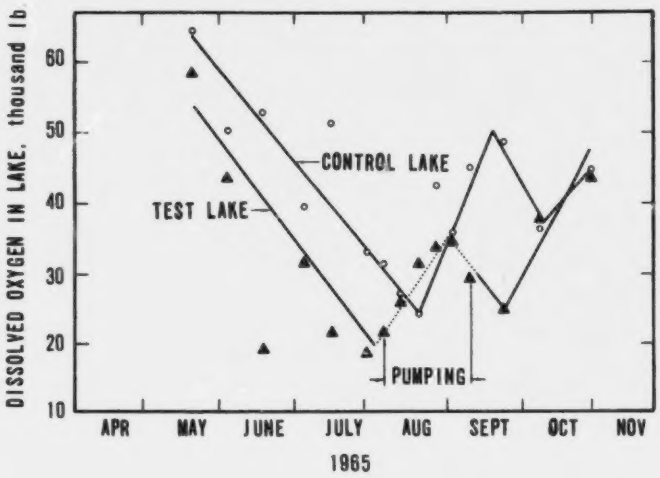

Figure 7. Total quantity of DO in test and control lakes.

reasonably high levels of DO at all depths, Figure 8 indicates further the undercapacity of this particular piece of equipment. It does show, however, that a favorable redistribution of DO resources is accomplished by artificial destratification.

Optimum Time for Destratification

Figures 5, 6, and 7 show a general deterioration in water quality from May to August. The total quantity of DO is lowered in the lakes, and the quantities of sulfide and manganese rise. While the mixing operation improved the water quality markedly in August, the goal of managing the DO resources in an impoundment through artificial destratification is to maintain quality throughout the summer season. The Boltz Lake studies were planned to begin in spring in an effort to prevent any water quality deterioration, but the lake was actually mixed in August because the mixing equipment was not ready for operation. Early season mixing was studied during 1966.[9]

Spring mixing has some hydraulic drawbacks: (1) The practical inefficiency of raising cold water when the volume of cold water is high (see Table 2), and (2) a naturally increasing stability with time at this season of the year (see the lower portion of Figures 1 and 4). Quality protection, not high hydraulic performance, is

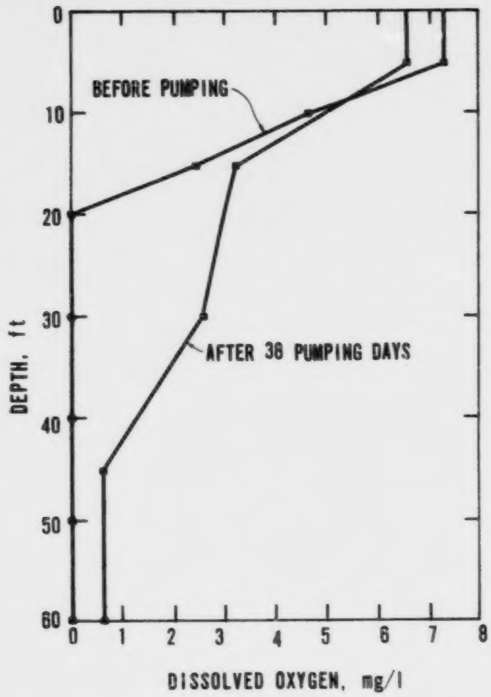

Figure 8. Redistribution of DO resources in Boltz Lake during artificial destratification.

the goal of this technique, however, and so some hydraulic sacrifices should be made in an attempt to prevent the quality deterioration that naturally occurs from the beginning of the spring season through the time of maximum stability in late July. Some of this poor performance could be overcome by pumping the warm water layer down into the water mass in the spring.

DISCUSSION

Hydraulic Performance

The data presented herein, and previously by the writers,[2,3] have indicated that artificial destratification is an effective method for improving water quality in impoundments, although to date (1967) it has been researched on only a relatively small scale. Any piece of mechanical artificial destratification equipment should be as hydraulically efficient as possible, and this efficiency can be evaluated by calculating the stability of the impoundment to be mixed. The calculated destratification efficiency during mixing, ''the net change in stability/the total energy input,'' is a good method of comparing one piece of destratification equipment with another, or evaluating various designs of a given piece of equipment. The equipment used in this study had a relatively poor destratification efficiency, 0.13 percent, probably because of poor hydraulic pump design.

Oxygenation Capacity

Effective water quality improvement is even more important than a high hydraulic performance; although the destratification efficiency of this pumping equipment was relatively low, the pump did supply 14,300 pounds of oxygen to the test impoundment with an expenditure of 14,300 kilowatt-hours of input energy. The calculated ''oxygenation capacity'' of this equipment was 1 pound of oxygen per kilowatt-hour. This is an acceptable value, although lower than other reaeration devices,[9] but this equipment was still undersized for Boltz Lake, as only enough oxygen was provided to satisfy the inorganic oxygen demand, and little DO was actually added to the water. In spite of this, the DO resources in the impoundment were favorably redistributed, and at least some DO was present at all levels of the lake after the mixing operation.

Best Time for Destratification

The major advantage of artificial destratification over devices for preventing escape of poor quality water from impoundments[1] is that mixing creates a twofold benefit – protecting downstream users and improving the quality of the water in the impoundment itself. To obtain these benefits, an impoundment should be destratified in the spring to prevent the formation of bad quality water rather than in midsummer to improve poor quality water. Spring mixing is necessary, in spite of the practical hydraulic inefficiency of raising cold water at this season of the year and the necessity of occasionally remixing during the summer season. These hydraulic sacrifices must be made in order to maintain good quality water throughout the heating season.

FUTURE PLANS

Continued study and research are necessary to completely define the potential of artificial destratification for water quality control. Comparisons of different destratification methods, spring mixing, and the mixing of larger bodies of water, must be made. A diffused air system was used, starting in the spring of 1966, for the destratification of Boltz Lake.[9] The disadvantage of this system is that it cannot be adapted to mixing warm water down in the spring. Eventually, other destratification methods must be tried, and water quality control by this technique must be demonstrated in a large storage reservoir.

CONCLUSIONS

Research on this project is continuing. The conclusions that may be drawn at this time (1967) are:

1. Engineering methods are available that will prevent the escape of poor quality bottom water from impoundments.

2. Artificial destratification is a better method of controlling water quality in impoundments, because it has the double benefit of preventing downstream damage and improving water quality in the impoundment itself.

3. The proper use of the stability concept for the calculation of the destratification efficiency of a mechanical mixing device is useful for comparing various pieces of equipment and improving equipment design.

4. In this study, reduced materials (sulfides and manganese) were eliminated from the test lake by the artificial destratification operation.

5. The equipment used in this study was insufficient in capacity to add the desired dissolved oxygen beyond the inorganic oxygen demand, but the available DO resources in the test lake were favorably redistributed by the pumping so that DO was present at all depths.

6. In spite of the practical hydraulic inefficiency of raising large volumes of cold water, spring mixing is better than midsummer destratification because early mixing will prevent any deterioration of bottom water quality.

ACKNOWLEDGMENTS

The writers wish to thank Lawrence Kamphake, George Holtzer, and Richard Shibiya for the laboratory chemical

analyses, and Glenn R. Gruber for help with the field work. In addition, acknowledgment is made of the cooperation of personnel of the State of Kentucky Department of Fish and Wildlife Resources.

Discussion of

"MANAGEMENT AND MEASUREMENT OF DO IN IMPOUNDMENTS"

Edward L. Thackston[*]

The authors present some useful suggestions for using the stability of an impoundment in calculating the hydraulic efficiency of artificial destratification devices. The use of destratification efficiency based on stability, however, is not without its pitfalls. It is to the authors' credit that they have been careful to point out the precautions to be taken, and the limitations and inadequacies of the concept.

The authors, however, have misinterpreted the mechanisms producing the typical temperature profile in stratified impoundments and the progression of stability throughout the summer. The original Figure 1 has two temperature profiles.

The dashed line marked 1 was stated to be the temperature profile that would result from solar heating if there were no wind, whereas the solid line marked 2 was stated to be the temperature profile that would be produced as the wind acted on profile number 1.

Although profile number 2 is an accurate representation of a typical realistic profile, profile number 1 is in error. If a reservoir is of uniform transparency, energy incident on the surface would be absorbed exponentially with depth, as described by Lambert's law, which states that

$$\frac{dI}{dL} = - KCI \qquad (2)$$

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in which I = the radiation intensity, L = the distance downward from the surface, C = the water color level, and K is a constant.

Integrating

$$I = I_o e^{-KCL} \tag{3}$$

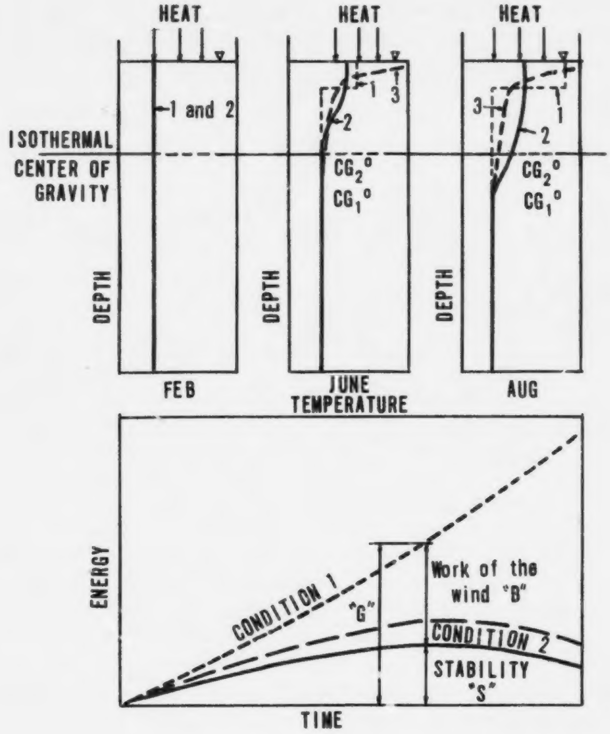
Thus, the profile that would be obtained in the absence of wind or surface cooling is really described by the line marked number 3, which has been added by the writer in Figure 9. The theoretical profile number 3 is modified to the actual profile number 2 by the action of wind mixing the upper layers, and also by convective mixing caused by the loss of heat from the surface to the atmosphere. This heat loss can occur by several mechanisms, as outlined by Raphael.[10] They include evaporative heat loss, conduction from water to air, and long-wave radiation to space.

The results of Harleman and Huber[11] indicate that convective mixing caused by surface cooling is the dominant mechanism involved in modifying profile 3 to profile 2. Their data were taken on an artificial indoor impoundment where no wind was present, but they showed the same typical changes.

Thus, the stability-time diagram shown in Figure 1 is misleading because it shows the stability labeled ''condition 1'' too high; the sharp thermocline depicted by temperature profile number 1 would never exist, even if no wind mixing were present.

The statement that, in the absence of wind mixing, the stability would increase throughout the summer is also incorrect. The stability will increase as the thermocline is lowered in the reservoir because of the continued solar heating and conductive heat transfer to the colder lower waters, only until the thermocline passes the isothermal center of gravity. From that time on, the stability will decrease, as more and more of the reservoir is included in the warm epilimnion. The maximum stability may never be reached in deep reservoirs, but it should in the shallow reservoirs described in the paper. Thus, the dashed line shown as ''condition 1'' should reach a maximum and trend downward also, even in the absence of any wind mixing.

The effect of both of these corrections to the ''condition 1'' line in Figure 1 is to lower it, and decrease


HEAT
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CG2°
CG1°
DEPTH
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DEPTH
FEB
JUNE
TEMPERATURE
AUG
ENERGY
CONDITION 1
"G"
Work of the
wind "B"
CONDITION 2
STABILITY
"S"
TIME

Figure 9. Corrected schematic diagram to demonstrate stability and work of the wind.

the distance marked ''work of the wind.'' In practice, the wind probably does not modify the temperature profile a great deal, except under unusual conditions such as violent storms. The amount of energy shown as ''work of the wind'' in Figures 1 and 3 is greatly overstated, and the maximum stability should be shown approximately by the dashed line marked 4 in Figure 9.

These comments do not affect the validity and usefulness of the stability concept, but they might affect thinking concerning natural stability trends at different times of the year, and thus affect decisions concerning mixing at different seasons.

Discussion of

"MANAGEMENT AND MEASUREMENT OF DO IN IMPOUNDMENTS"

*C.H.J. Hull**

Impoundments are becoming increasingly important in the management of water resources. Long used for regulating the quantity of water available for beneficial purposes, reservoirs have been legislatively recognized as a means of regulating the quality of water since 1961, when Public Law 87-88 was passed. It was inevitable that this should lead to increased research in the effects of impoundments on water quality, not only in regulated streams, but in impoundments themselves. Symons and his co-workers have contributed significantly to this research effort, as well as to the understanding of stratification of impounded waters and its artificial control. The writer hopes to add a small measure of understanding and improvement to the approach taken by the authors in their latest of a series of papers on this subject.

The authors have defined the difference between the stability curves for ''condition 1'' and ''condition 2'' in Figure 1 as the ''work of the wind,'' citing Hutchinson[7] as the source of this definition. There are natural vertical mixing forces other than the wind at work in reducing the stability of many-probably most-impoundments and lakes. For example, the ebullition of anoxic gases from bottom deposits is a fairly common phenomenon. These natural gases, rising through the water to the surface, cause vertical mixing in the same way that air bubbles introduced artificially cause vertical mixing. This natural phenomenon has been studied at length by Mortimer [12,13] and Ohle,[14] as well as others.

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*Head, Program Planning Branch, Delaware River Basin Commission, Trenton, New Jersey.

Bradley[15] has investigated a different type of natural vertical density current, and has inferred from field observations and laboratory experiments that vertical density currents are responsible for moving particles from the surface waters of lakes to the bottom rapidly–much more rapidly than can be accounted for by settling according to Stokes law. He suggested that these vertical currents could be induced by the greater density of surface water in which particles are brought close together, by some means, so that they and the water containing them will together be heavier per unit volume than water with less sediment, causing the particle-laden water to sink vertically. Bradley reasoned that such vertical density currents should be expected in any lake that produces calcite particles from either photosynthesis of phytoplankton or warming of the surface water, and in any lake that produces dense populations of microorganisms whose dead remains have specific gravities greater than the specific gravity of the water in which they lived.

In considering the limnological significance of vertical density currents, Bradley[15] concluded that such currents should carry with them appreciable amounts of heat and dissolved oxygen (DO) into the deeper parts of a lake. He pointed out that such a transfer of oxygen could seriously affect calculation of the rate of oxygen consumption in the hypolimnion. Similarly, they could influence calculations of stability as related to wind and other natural forces on the one hand, and artificial mixing devices on the other. From these considerations, the writer is led to suggest that ''work of nature'' would be technically more precise than ''work of the wind'' to designate the difference between the stability curves for ''condition 1'' and ''condition 2'' in Figure 1.

Discussion of
"MANAGEMENT AND MEASUREMENT OF DO IN IMPOUNDMENTS"
Authors' Closure

The writers wish to thank Thackston and Hull for their interesting comments.

The writers accept as valid the suggestions of Thackston and Hull concerning the natural mechanisms for the distribution of heat that operate in a reservoir. The authors agree that Hutchinson's[7] term ''work of the wind'' is somewhat misleading. What was really meant in that section of the paper was that much of the incoming heat is distributed vertically by natural phenomena and that the calculated stability is a measure of the non-uniformity of the natural distribution of heat. The energy represented by the stability then would have to be provided by mechanical equipment, if a reservoir were to be completely mixed.

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PAPER NO. 10

CALCULATION TECHNIQUE FOR DESTRATIFICATION EFFICIENCY

J.M. Symons and G.G. Robeck

This paper, the fourth in the series of eight concerning artificial destratification for reservoir water quality control, presents, in detail, one method for calculating the "destratification efficiency" (DE) for any mechanical mixing device. The use of DE has been previously proposed for evaluation of mixing equipment performance (Papers No. 8 and 9). This paper was not published.

The correct reference citation is:

Symons, J. M. and G. G. Robeck. "Calculation Technique for Destratification Efficiency." Unpublished Mimeographed Report. November 1966. 9 pp.

As artificial destratification of lakes and reservoirs has become more popular as a method of maintaining good water quality, the need has arisen for comparing the effectiveness of various pieces of mechanical equipment used for destratification. Although the goal of any mixing operation is to maintain or improve water quality, this improvement should be accomplished as efficiently as possible. Therefore, some arbitrary index of mechanical performance is needed.

Previous papers[1,2] have proposed the use of stability changes as one method of calculating an index of destratification performance. Stability has long been used by limnologists[3] as an arbitrary measurement of the intensity of stratification in any body of water. Stability is calculated in work or energy units, and is defined as the

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work that must be done to the water (the energy gained) to lift the entire weight of a body of water the vertical distance between the center of gravity when the body of water is in a given state of stratification and the center of gravity when the water body is isothermal. Stability also may be thought of as the minimum energy required to completely mix a stratified body of water.

The proposed index of mechanical performance, ''destratification efficiency'' (DE), is defined as:

$$DE = \left[\left(\frac{\text{net change in stability from } t_1 \text{ to } t_2}{\text{total mechanical energy input from } t_1 \text{ to } t_2}\right) \times 100\right] \tag{1}$$

where t_1 and t_2 are the times of starting and stopping mixing, respectively. Although this index of performance has its limitations,[1,2] it also has its utility. This report describes in detail how the *DE* may be calculated.

For this example, data taken in the summer of 1966 in Dodgeville, Wisconsin, are used. The calculations are shown in Table 1. First the lake is divided into 5-foot vertical layers, as shown in Column 1. Then, from a volume-depth profile, the volume in acre-feet is found for each layer (Column 2). The arithmetic sum of Column 2 should be the total volume of the lake. Next, on the assumption that the body of water is isothermal and at 4°C, the weight of each layer in millions of pounds is calculated, using a specific gravity of 1.00000. One acre-foot of water weighs 2.719×10^6 pounds at a specific gravity of 1.00000. These weights are shown in Column 3.

Next, to calculate the depth of the isothermal center of gravity, moments are taken about the surface of the lake. To do this, the weight of each layer is multiplied by the distance from the surface of the lake to the depth of the center of the layer. For example, in Table 1, the weight of the first layer, $1{,}163.7 \times 10^6$ pounds, is multiplied by 2.5 feet (the distance from the surface to the depth of the center of the layer) to yield $2{,}909.3 \times 10^6$ foot-pounds. This number is then entered in Column 4. To obtain the depth of the isothermal center of gravity, the arithmetic sum of Column 4 is divided by the arithmetic sum of Column 3. In this example, the depth of the isothermal center of gravity is calculated as 8.6888 feet. Usually, four decimal places are needed in the calculation of the depth of the isothermal center of gravity.

Table 1. EXAMPLE OF CALCULATION OF DESTRATIFICATION EFFICIENCY (DE)

Layer, ft	Volume, ac-ft	Weight (specific gravity = 1.00000), 10^6 lbs	Moments about surface, 10^6 ft-lbs	Distance center of layer to isothermal center of gravity, ft	Potential energy lost (-) or gained (+) moving layers to isothermal center of gravity, 10^6 ft-lbs	Multiplying factor, 10^6 ft-lbs	Temperature profile 6/30/66, °C	Specific gravity factor see Fig. 1, dimensionless	Difference in energy lost or gained for each layer, 10^6 ft-lbs	Temperature profile 7/18/66, °C	Difference in energy lost or gained for each layer, 10^6 ft-lbs
(1)	(2)	(3)	(4)	(5)	(6)	(7)	(8)	(9)	(10)	(11)	(12)
0-5	428	1,163.7	2,909.3	-6.1888	-7,201.9	-7,200	28.5	-0.00390	+28.1	24.9	+21.0
5-10	328	891.8	6,688.5	-1.1888	-1,060.2	-1,100	24.1	-0.00270	+3.0	24.4	+3.1
10-15	228	619.9	7,748.8	+3.8112	+2,362.6	+2,400	15.3	-0.00093	-2.3	23.2	-6.0
15-20	136	369.8	6,471.5	+8.8112	+3,258.4	+3,200	11.5	-0.00042	-1.3	22.4	-7.4
20-25	60	163.1	3,669.8	+13.8112	+2,252.6	+2,300	9.8	-0.00026	-0.6	20.8	-4.5
25-28	8	21.8	577.7	+17.8112	+388.3	+400	8.6	-0.00016	-0.1	18.7	-0.6
	1,188	3,230.1	28,065.6		-0.2	0			+26.8		+5.6
									Stability = 10.1 kw-hrs		Stability = 2.1 kw-hrs

Isothermal center of gravity:

$$\frac{28,065.6}{3,230.1} = 8.6888 \text{ feet from surface}$$

$$DE = \frac{(10.1 - 2.1)}{(6.2)(24)(18)} \times 100 = 0.30\%$$

Column 5 shows the distance in feet from the depth of the isothermal center of gravity to the depth of the center of each layer. For example, for the first layer, 2.5000 feet (the depth of the center of the first layer) is subtracted from 8.6888 feet to obtain -6.1888 feet, which is the first number in Column 5. All distances above the isothermal center of gravity are given a minus sign, whereas the distances below the isothermal center of gravity are given a plus sign.

Next, a check is made to show whether or not the stability would be equal to or near zero if the lake were isothermal. This is done by calculating the potential energy lost when layers above the depth of the isothermal center of gravity are moved down to that depth, and the potential energy gained when layers below the depth of the isothermal center of gravity are moved up to that depth. For example, for the first layer, moving a weight of 1,163.7 x 10^6 pounds a distance of -6.1888 feet from the depth of the center of the first layer to the depth of the isothermal center of gravity yields a loss in potential energy of -7,201.9 x 10^6 foot-pounds. This number is the first number in Column 6. All the numbers in Column 6 are obtained by multiplying the appropriate number in Column 3 times the appropriate number in Column 5, keeping careful track of the algebraic sign. If the arithmetic to this point is correct, the algebraic sum of Column 6 should be very close to zero, because an isothermal body of water, as this one is presently assumed to be, should have zero stability. As shown in the example, the algebraic sum of Column 6 is -0.2 x 10^6 foot-pounds, which is close enough to zero.

Column 6 shows that if this body of water were isothermal, and at 4°C, the potential energy lost or gained moving each layer to the depth of the isothermal center of gravity would be as shown in Column 6, and would balance. The fundamental concept for calculating the stability for any other temperature profile than the one used for calculating Column 6 (a uniform 4°C temperature throughout the depth) is to calculate the *difference from the Column 6 value* in the potential energy lost or gained to move a given layer to the depth of the isothermal center of gravity, when that layer has a different temperature from 4°C and, therefore, a different weight.

For example, if the first layer in this lake were at 25°C, instead of 4°C as originally assumed, it would be somewhat lighter than the 1,163.7 x 10^6 pounds shown in Column 3. Therefore, the potential energy lost by moving this layer down to the depth of the isothermal center of gravity would be correspondingly *less* than the -7,201.9 x 10^6 foot-pounds shown in Column 6. Rather than calculate the actual weight of any layer at its actual temperature, calculation of the *change* in weight for any layer at its actual temperature is preferred, although either technique is acceptable. The *difference in weight* method yields the difference in potential energy directly, and this is the desired stability value.

Figure 1 is used to calculate this difference. This Figure shows the specific gravity factor (specific gravity at given temperature - 1.00000) for water temperatures in the range 5° to 30°C. For example, at 25°C, the specific gravity of water is 0.99707 and the difference between the specific gravity at 25°C and 4°C is -0.00293. Because the numbers in Column 6 were calculated for 4°C water, the specific gravity factor from Figure 1 may be used to calculate the difference in potential energy gained or lost for this layer when it is at 25°C instead of 4°C.

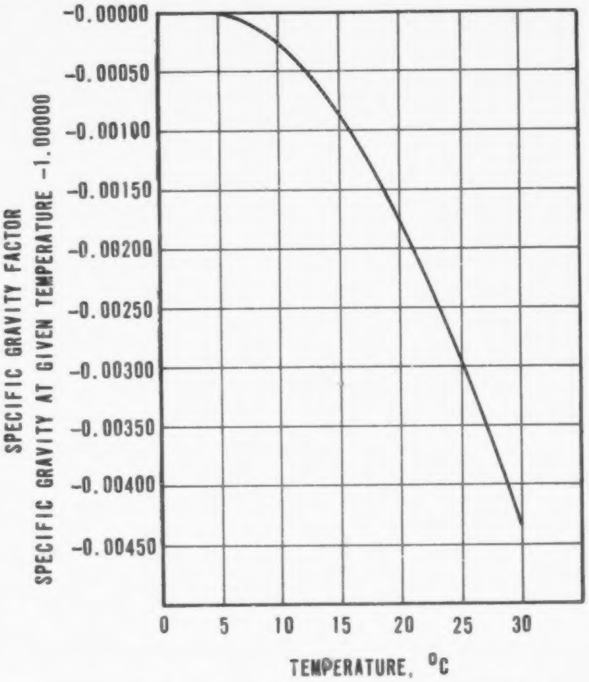

Figure 1. Specific gravity factor for certain water temperatures.

The following equation* approximating the curve shown in Figure 1 has been developed through a computer application of the least squares technique:

$$\begin{aligned}&\text{Specific Gravity Factor} = \\ &- 0.2260569\ (10^{-5}) + 0.1546919\ (10^{-5})\ T \\ &+ 0.2141968\ (10^{-5})\ T^2 - 0.6508630\ (10^{-6})\ T^3 \\ &+ 0.1975524\ (10^{-7})\ T^4 - 0.1894802\ (10^{-9})\ T^5 \qquad (2)\end{aligned}$$

where T = temperature in °C.

For this example, the temperature profile in this lake (temperature was measured at the center of each layer) on June 30, 1966, is shown in Column 8. The specific gravity factor taken from Figure 1 is shown in Column 9. The difference in potential energy gained or lost that was caused by the temperature of a given layer being different from 4°C is shown in Column 10 and is obtained by multiplying Column 9 by Column 7. Column 7 values are the Column 6 values rounded off to two significant figures. This is sufficiently accurate for the difference method, and eases calculations. The Column 7 values should be adjusted to have an algebraic sum equal to zero.

For example, for the first layer, $-7,200 \times 10^6$ foot-pounds is multiplied by -0.00390. This yields a difference in potential energy lost for the first layer of $+28.9 \times 10^6$ foot-pounds. Similar calculations are made for each layer and the algebraic sum of the difference in potential energy gained or lost for each layer is the stability of the lake at this time, in millions of foot-pounds. Multiplying this algebraic sum by 3.77×10^{-7} converts the stability to kilowatt-hours. In this example, the stability on June 30, 1966, was 10.1 kilowatt-hours.

The stability calculation 18 days later is shown in Columns 11 and 12. Here, the specific gravity factor from Figure 1 was not entered, as it need not be permanently recorded. The stability on July 18, 1966, was calculated to be 2.1 kilowatt-hours. The destratification efficiency for this mechanical device during the 18-day period of operation is calculated by dividing the *net* change in stability (10.1 – 2.1) by the total mechanical energy input during the 18 days. This calculation assumes that during the 18-day operation the stability of the lake would not have increased if the mixing device had not been in operation. Actually, at this season of the year, the stability of lakes in general is rising, so if a control–a nearby unmixed lake of similar physical size and shape–had been available for this operation, any change in its stability would be added or subtracted to obtain the *net* change in stability from t_1 to t_2 for use in Equation (1).

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*The authors are indebted to Mr. Joseph W. Dewitt, Civil Engineer, U.S. Army Corps of Engineers District Office, Savannah, Georgia, for providing this equation.

If a zero change of stability in a control were assumed, this piece of mechanical equipment would have a destratification efficiency of:

$$\left[\left(\frac{8.0}{2678}\right) \times 100\right] = 0.30\%$$

The denominator of the fraction is calculated by multiplying the power of the motor, 6.2 kilowatts, times the time of operation, 18 days. This yields a total energy input of 2,687 kilowatt-hours.

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PAPER NO. 11

WATER QUALITY IN IMPOUNDMENTS AND MODIFICATIONS FROM DESTRATIFICATION

W.H. Irwin, J.M. Symons, and G.G. Robeck

This paper, the fifth in the series of eight on artificial destratification for reservoir water quality control, introduces the data from the summer of 1966. In 1966, the same lake that was mixed in 1965 was mixed again, but this time with a diffused-air pump instead of a mechanical pump. Also in 1966, the lake was mixed several times during the summer, instead of once as in 1965. Finally a second, larger lake was also mixed in 1966. In this paper, calculated summer-long averages for 18 water quality parameters for various layers of the lakes are presented to show the influence of mixing on quality. Also, a method is proposed for estimating the flow discharge of artificial destratification devices. Finally, the results of a bioassay study of the potential toxicity of sulfide-laden hypolimnetic water are also included.

The correct reference citation is:

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INTRODUCTION

Regulation of streamflow to provide dilution water to subsequently prevent downstream deterioration of water quality received an impetus in the 1961 amendments to the Federal Water Pollution Control Act. Because impoundments would be the source of water for any streamflow augmentation, the nature of impounded water quality becomes important. In 1962 the Engineering Research Activities group at the Federal Water Pollution Control Administration's Cincinnati Water Research Laboratory began a study of impoundment behavior as it relates to streamflow regulation. These studies included laboratory-scale investigations of water quality behavior, as well as field studies on artificially overcoming thermal stratification to improve the quality of impounded water. Publications resulting from this project include a literature review,[1] and reports of an impoundment water quality behavior study[2] and of several artificial destratification studies.[3,4,5,6] The purpose of this paper is to briefly review the influences of impoundments on water quality and to discuss the results of the artificial destratification research.

WATER QUALITY IN STRATIFIED IMPOUNDMENTS

The quality of impounded waters varies from watershed to watershed, and even within watersheds, depending on soil population density, man's activities, and meteorological conditions.[7,8,9,10] Quality will also vary with the shape of the reservoir basin, the degree of protection from wind, and the amount of water exchange. Because of these variables, generalizations can be made, but predictions of specifics of water quality are difficult.

Water quality in an impoundment is often better than that of the original stream. Some factors that improve quality are: 1) blending and diluting of influent water with impounded water to reduce the effect of peak concentrations of pollutants, 2) detention and lower velocity in the impoundment to permit sedimentation and precipitation of solids, 3) exposing the water to light and air for longer periods to permit greater biodegradation of organic wastes, and 4) higher trophic levels resulting from the transfer of energy to protoplasmic units during and following biodegradation. The subsequent transfer and concentration of energy and protoplasm into higher levels in the food chain will result in increased available fish food, unless the impoundment waters have reached a eutrophication level beyond optimum.

In contrast to the beneficial influences noted, water quality in impoundments is often lowered after it thermally stratifies. For example: 1) thermal stratification results in a reduced heat budget for the entire lake, 2) heat is concentrated near the surface, causing more evaporation and greater heat loss to the air, and 3) because the cold layers are usually below the photosynthetic zone and are isolated from the oxygen supply in the air by the warm layers over them, algal or atmospheric reaeration does not occur. Oxygen in this zone is soon depleted and chemical stratification begins.

As a result of the exhausted oxygen supply, reductive products causing quality degradation accumulate in the zone that is now septic. Degradation effects result from: 1) anaerobic organisms building populations in balance with their food supply and formation of methane, hydrogen sulfide, and ammonia nitrogen,[7,9] and 2) under reducing conditions, iron and manganese that settle from the aerobic zone in an oxidized insoluble form going back into solution.[7]

The plankters in the warm upper waters produce sizable populations that eventually die and create tons of organic materials that sink into lower layers of the impoundment, thereby furnishing more substance for oxygen demand and chemical reduction. The warmer temperatures and longer periods of stratification in the southern United States further aggravate this quality deterioration. The entire process continues until the thermal and chemical stratification is broken artificially or naturally.

IMPOUNDMENTS STUDIED

The impoundments studied are located in northern Kentucky. Earthen dams across ravines form deep-sided basins that are generally narrow, relatively long, and protected from strong winds. Pollution from factories and cities is minor. The watersheds are only lightly fertilized, except for a few acres used for tobacco farming. The waters are clear and clean, but mildly eutrophic. Algal crops are present, but rooted aquatic plants are scarce because of the large percentage of deep water. Fair to good fish crops grow in the reservoirs, which are frequented by fishermen.

The lakes stratify thermally and chemically each year. In northern Kentucky impounded water begins to absorb heat by early March, and by mid-April strong thermal stratification may be present. The shearing plane, a plane between great density differences,[7] remains at 15 feet or less from the surface during most of the summer and early fall, varying downward slightly as the season progresses. Lakes more protected from the wind in this area can have the shearing plane located as shallow as 10 feet from the surface. The cooler air of late September and October removes heat from the upper water, causing the plane of greatest-density-change to descend, until the entire water becomes isothermal before or during early November.

Two lakes were studied in 1965 and 1966 and a third lake in 1966. Bullock Pen Lake, with 142 surface acres, 48 feet maximum depth, and 3,200 acre-feet volume, was not mixed during either year, but remained as a normal stratified reservoir, thereby acting as a control. Boltz Lake, with 96 surface acres, 62 feet maximum depth, and 2,900 acre-feet volume, was mixed once during 1965 and four times during 1966. Falmouth Lake, with 225 surface acres, 42 feet maximum depth, and 4,600 acre-feet volume, was mixed five times during 1966.

APPARATUS. ANALYTICAL AND EXPERIMENTAL METHODS

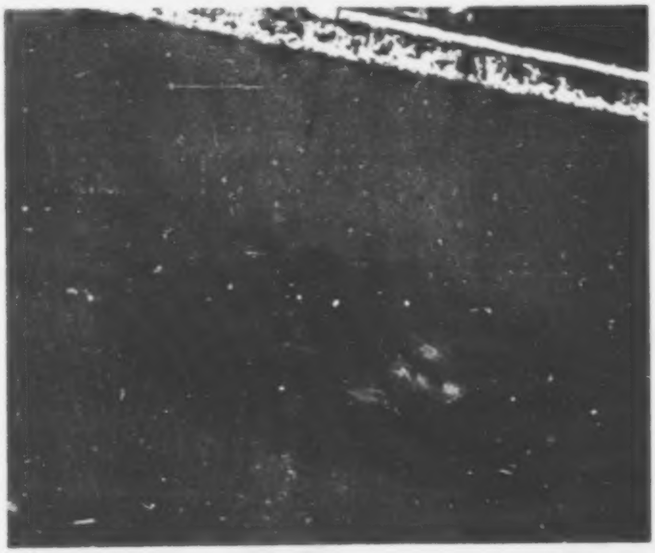
During 1965, Boltz Lake was artificially destratified by pumping cold water from the bottom to the surface, where it gained heat and lost density on contact with the epilimnetic water. A mixed-flow pump with a suction pipe attached below the impeller and extended downward to just above the lake bottom, was mounted on a float anchored over the deepest part of the lake. A strainer with a sheet-metal bottom was attached to the lower end of the suction pipe to admit water from the sides only, thus avoiding pumping bottom mud and ooze.

The 12-inch-diameter mixed-flow pump was powered by a 26-horsepower gasoline engine operating at 2,000 rpm. The output of the gasoline engine at this speed was about 21 horsepower. Pumping against the low elevation head, this pump had a capacity of about 13 acre-feet per day (4.2 mgd, 6.4 cfs, 2,880 gpm) and a water discharge velocity of about 8.4 feet per second. Nothing was gained by moving the pump; the entire lake volume could be mixed by pumping at a single position over the deepest water.[3]

In 1966 a different mechanical system was used. A 30-horsepower portable, electrically-driven rotary air compressor was attached to a system of diffuser stones and air tubes that were designed to carry the compressed air to the lake bottom where it could be released effectively. To ensure good distribution of the air to each diffuser stone, an orifice was placed in each hose to create some head loss. Thus, each diffuser received approximately the same volume of air, even if the diffuser apparatus was not level on the uneven lake bottom. A flow meter was attached to the outflow line of the compressor to measure the quantity of air pumped.

The average rate of air released at the 62-foot depth in Boltz Lake was about 112 cfm, and at the 42-foot depth in Falmouth Lake, about 115 cfm. The compressor was operated from the shore with a plastic pressure hose leading to 16 diffusers that were arranged in a cross pattern group on the lake bottom. The air was released in a 40-foot diameter circular pattern. The influent hose was weighted and submerged in order to leave the entire lake surface free for recreational use.

When the compressor was started, the air passed through the hose and diffuser stones into the cold bottom water of the lake. As the air rose to the surface, water was carried upward also. Above the diffusers at the lake surface a large boil of water with small air bubbles was formed. The upward flow of water raised the surface in the boil area to 4-5 inches above the lake surface. The boil diameter was about 100 feet. The color aerial photograph shows the discharge pattern of the diffused air. The algae-free bottom water is contrasted against the surface water that contains algae. The four arms of the diffuser system can be seen.



To determine the influence of stratification and destratification on water quality, the lakes were sampled periodically during the 1965 and 1966 summer seasons. Samples were taken at various depths, near the mixing site in the two destratified lakes, and near the dam in the control lake. These water samples were analyzed for temperature, and conductivity, and for concentrations of dissolved oxygen (DO), chemical oxygen demand, alkalinity, total solids, sulfates, sulfides, iron, manganese, calcium, magnesium, the four nitrogen forms, total hydrolyzable phosphorus, and soluble hydrolyzable phosphorus. In addition, algal samples were taken and are currently (1967) being analyzed. The influence of artificial destratification on algal populations will be reported in the near future.

Temperature, conductivity, and concentrations of DO, and alkalinity were determined in the field. The remaining analyses were performed in the chemical laboratory of the Engineering Research Activities group. In general, all analyses were made in accordance with *Standard Methods*,[11] with some modifications to permit use of a Technicon AutoAnalyzer.*

RESULTS

Chemical

During 1965, Boltz Lake was mixed once during August and early September and in 1966, it was mixed once each in early June, mid-June, mid-July, and mid-August. In 1966, Falmouth Lake was mixed once each in mid-May, mid-June, early July, late July, and in early August. It is impossible in this brief paper to present all the chemical water quality data. Some of the 1965 data, however, have been detailed in References 4 and 5; the 1966 data will be presented in the near future.[6]

This paper attempts to summarize these data through the use of calculated averages to show some of the major effects of artificial destratification on water quality. The concentrations of most constituents varied with time and with depth. While the use of calculated averages has certain inherent disadvantages, the averages are nevertheless helpful in highlighting which water quality characteristics were changed by artificial stratification and which were relatively unaffected.

Table 1 shows the percentage gained or lost for given water quality parameters during the 1966 mixing operations. To obtain these data, the weighted vertical average concentrations (total weight in the lake divided by total lake volume) from water samples taken just before and just after each mixing were averaged for the entire summer

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*A product of Technicon Corporation, Tarrytown, N.Y.

Table 1. INFLUENCE OF ARTIFICIAL DESTRATIFICATION ON VARIOUS WATER QUALITY CHARACTERISTICS IN TEST LAKES - 1966

Characteristic (1)		Boltz Lake ''Before'' average (2)	Boltz Lake ''After'' average (3)	Boltz Lake % gain (+) or loss (-) (4)	Falmouth Lake ''Before'' average (5)	Falmouth Lake ''After'' average (6)	Falmouth Lake % gain (+) or loss (-) (7)
Heat	BTU/ac-ft	79×10^6	92×10^6	+17	80×10^6	94×10^6	+17
Dissolved Oxygen	mg/l	7.6	7.3	-4	3.1	3.0	-3
Chemical Oxygen Demand	mg/l	16	13	-18	29	28	-3
Iron	mg/l	0.16	0.12	-25	0.52	0.38	-27
Manganese	mg/l	0.51	0.38	-25	0.08	0.10	+25
Sulfide	mg/l	0.14	0.04	-71	1.44	0.20	-86
Total Solids	mg/l	177	186	+5	203	204	±0
Conductivity	micromhos	270	275	+2	312	267	-14
Alkalinity	mg/l as $CaCO_3$	130	128	-2	173	177	+2
Calcium	mg/l as $CaCO_3$	87	82	-6	92	92	±0
Magnesium	mg/l as $CaCO_3$	48	46	-4	58	61	+5
Sulfates	mg/l	31	31	±0	24	25	+4
Ammonia Nitrogen	mg/l N	0.19	0.24	+31	0.08	0.04	-50
Nitrite Nitrogen	mg/l N	0.02	0.03	+50	0.02	0.01	-50
Nitrate Nitrogen	mg/l N	0.20	0.26	+30	0.21	0.13	-38
Organic Nitrogen	mg/l N	0.35	0.60	+71	0.32	0.54	+69
[a]Soluble Hydrolyzable Phosphorus	mg/l P	0.07	0.04	-43	0.51	0.43	-16
[a]Total Hydrolyzable Phosphorus	mg/l P	0.12	0.08	-33	0.62	0.52	-16

Note: Method of calculation explained in text

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[a] Hydrolyzable phosphorus has an operational definition. It is the phosphorus that is hydrolyzed to orthophosphate during 15 minutes heating in an autoclave at 15 psig under acid conditions. This procedure measures all of the orthophosphate, all of the polyphosphate and an indeterminate amount of the organic phosphate in the original sample. Soluble hydrolyzable phosphate is determined from a filtered sample.

season. These data are shown in Columns 2, 3, 5, and 6. A comparison of these ''before'' and ''after'' averages shows the percent gained or lost by a given water quality characteristic as caused by artificial destratification.

As an example of this technique, the method of calculation used for the sulfide data in Falmouth Lake is shown graphically in Figure 1. Here, the decrease in the vertical weighted average sulfide concentrations during each of the five Falmouth Lake mixings is shown. The five before-mixing data points (solid dots) were averaged, as were the five after-mixing data points (open circles). The difference between the before and after average values, divided by the before average value is the percent decline in sulfides caused by mixing.

Each mixing operation increased the heat content of each lake an average of about 17 percent. Mixing transferred the surface heat to the depths of the lake, eliminated the cold water, cooled the surface water, and permitted the surface to absorb more radiant energy.

The DO content decreased slightly in both lakes during mixing; however, oxygen was transferred into the water, as evidenced by a decrease of COD, iron, and sulfides in both lakes and manganese in Boltz Lake. The reason the manganese content rose in Falmouth Lake is not known at this time (1967).

Apparently the reaeration capacity of the mixing operation was not sufficient to remove the oxygen-demanding

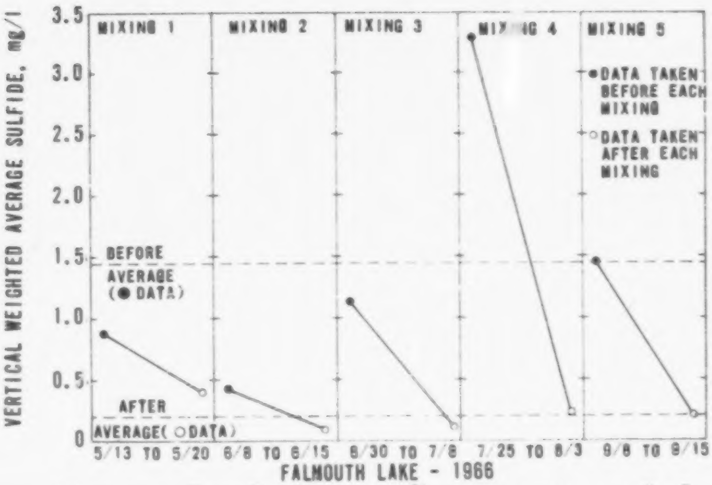

Figure 1. Example data to demonstrate method used to obtain data in Table 1.

materials as well as increase the DO concentration in the water. Calculating the respective oxygen quantities transferred through the water from the diffused air bubble and from surface reaeration is difficult. An attempt was made to analyze the gas content of the rising air bubbles, but the oxygen and nitrogen contents of these bubbles were about the same as found in air. Carbon dioxide and methane were detected in these rising gas bubbles, however.

The inorganic content of the lakes, as represented by total solids, conductivity, alkalinity, calcium, magnesium, and sulfates, was influenced very little by the mixing operation. In all but one instance the percentage changes were within ± 10 percent.

Boltz Lake showed increases in ammonia-, nitrite-, and nitrate-nitrogen content, while Falmouth Lake showed decreases for these same three constituents. The organic nitrogen content of both lakes went up, while the total and soluble hydrolyzable phosphorus concentrations went down. At this time (1967) the reason for these changes is not known, but as shown in columns 2, 3, 5, and 6 of Table 1, both the before and after average concentrations for these six constituents were low. Therefore, even though mixing did cause some percentage increases in the nutrient content of the two test lakes, the increases in concentrations were small and did not cause algal blooms.[4]

As mentioned before, averages can be misleading because of variation in concentrations that occur with time during the summer season and with depth in the lake. To show the vertical distribution of the water quality characteristics under study, summer season averages of each parameter were calculated for several depths in each of the three lakes and are shown in Figures 2-18. To provide a complete picture of vertical distribution, the data for the control lake (Bullock Pen Lake) for both 1965 and 1966 are included along with the 1965 (one mixing) and 1966 (four mixings) Boltz Lake data and the 1966 (five mixings) Falmouth Lake data. For each lake, the summer average for the entire lake is also included for comparison purposes.

Figures 2 and 3 show that both heat content and DO are vertically distributed with the highest concentrations being near the surface of the lake. Figures 5, 6, 7, 9, 13, 17, and 18 show that iron, manganese, sulfides, conductivity, ammonia nitrogen, and total and soluble hydrolyzable phosphorus were vertically distributed in the lakes, with maximum concentration being near the bottom. Figures 4, 10, 11, 12, 14, 15, and 16 show that chemical oxygen demand (COD), total solids, calcium, magnesium, sulfate, nitrite nitrogen, nitrate nitrogen, and total organic nitrogen were relatively uniformly distributed in the lake from top to bottom. Because water quality problems in a stratified body of water are often localized in a vertical direction, recognition of the vertical distribution of water quality characteristics must be considered when assessing the value of artificial destratification, in addition to the data given in Table 1.

Biological

Although bottom water is lethal to fish and fish food because of the lack of oxygen, the possibility of its being toxic, even after reaeration, has not been investigated previously. Therefore, during the 1965 and 1966 mixing operation a satellite study was performed to determine whether destratification by reaeration of the bottom water of stratified impoundments would prove harmful to fish crops.

In the summer of 1965 bioassay toxicity studies involving the waters of the hypolimnion of Bullock Pen Lake were made. Initially, two 100-gallon steel tanks were used; one received water from near the lake surface, the other from near the lake bottom. The water flow was continuous at about 1 gallon per minute to each tank. The bottom water was aerated in its bioassay tank to about 4 to 5 milligrams of DO per liter using an air pump and diffuser stones. Fish from Bullock Pen Lake were used as specimens.

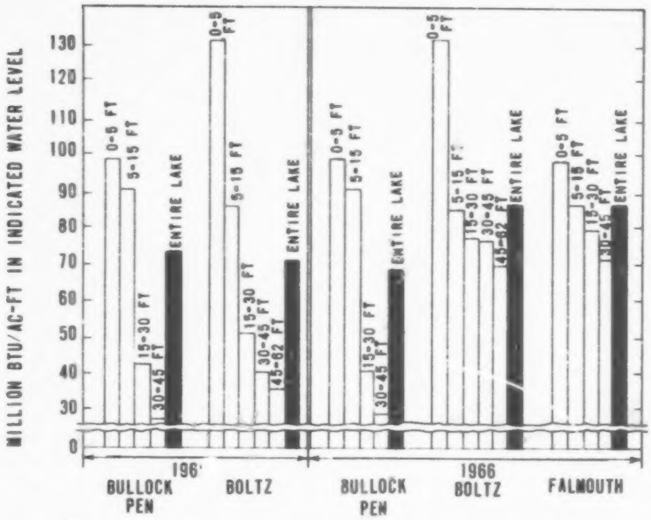

Figure 2. April and October average heat content for each layer and entire volume for each lake.

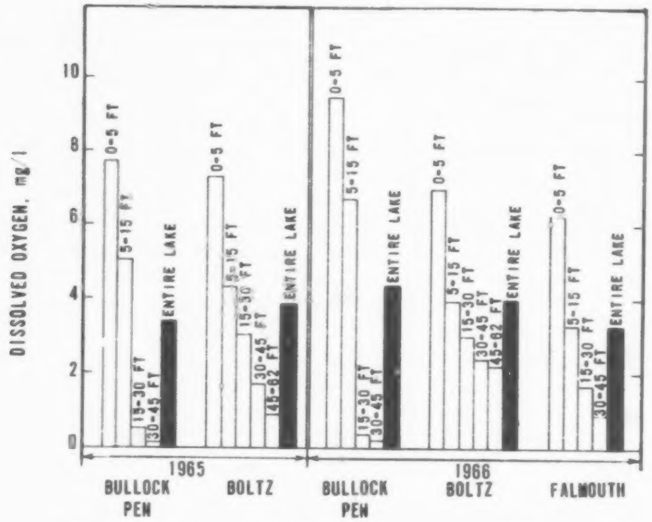

Figure 3. April to October average DO concentration for each layer and entire volume for each lake.

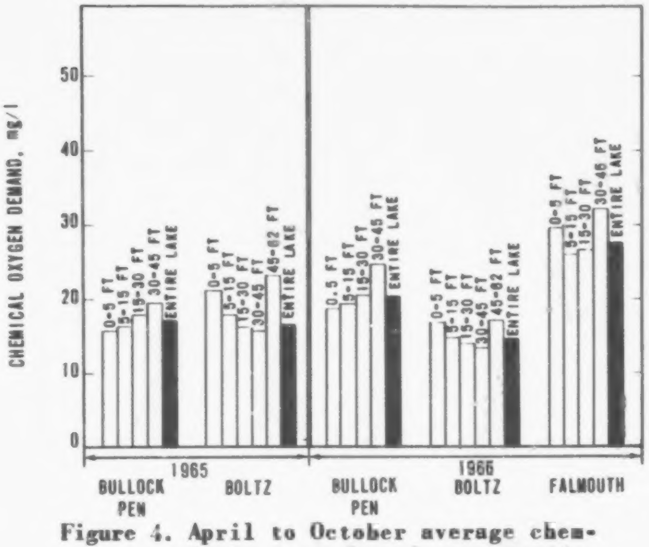

Figure 4. April to October average chemical oxygen demand concentration for each layer and entire volume for each lake.

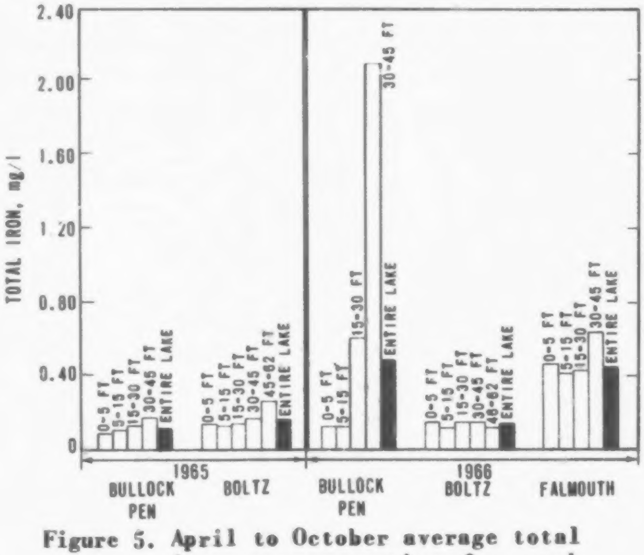

Figure 5. April to October average total iron concentration for each layer and entire volume for each lake.

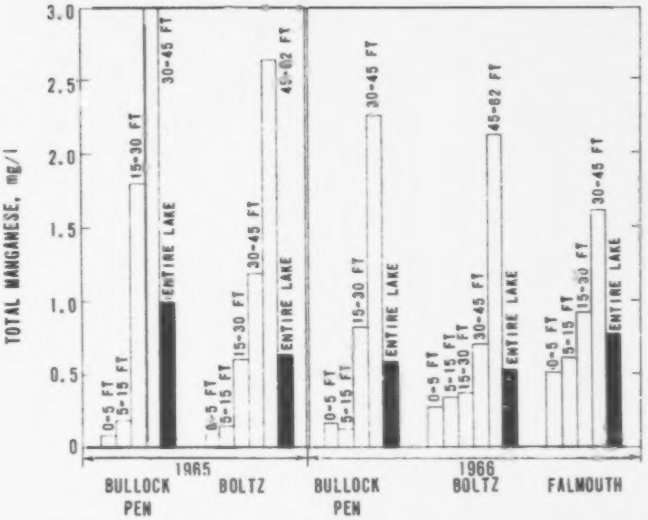

Figure 6. April to October average total manganese concentration for each layer and entire volume for each lake.

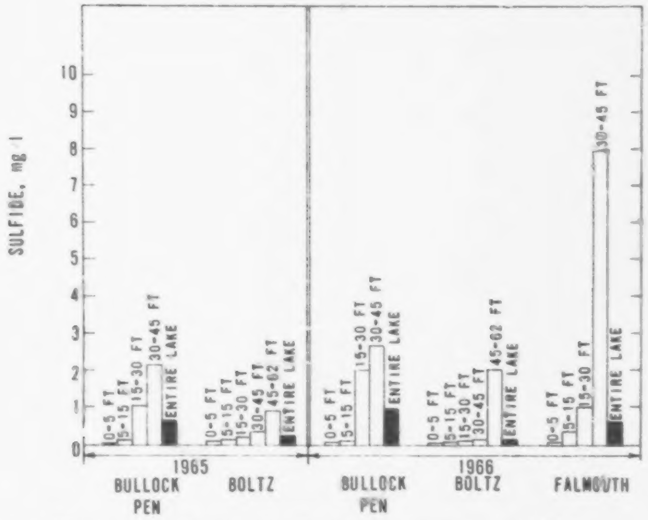

Figure 7. April to October average sulfide concentration for each layer and entire volume for each lake.

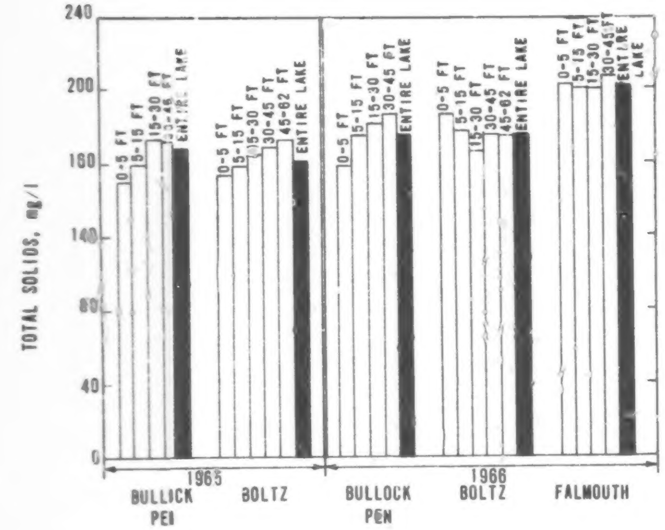

Figure 8. April to October average total solids concentration for each layer and entire volume for each lake.

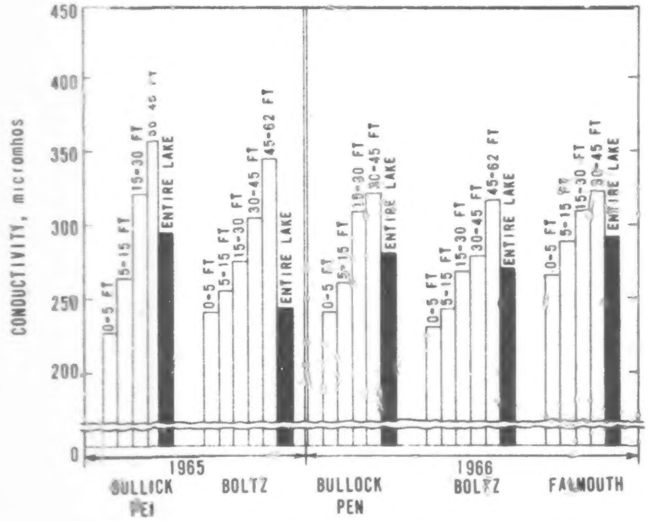

Figure 9. April to October average conductivity for each layer and entire volume for each lake.

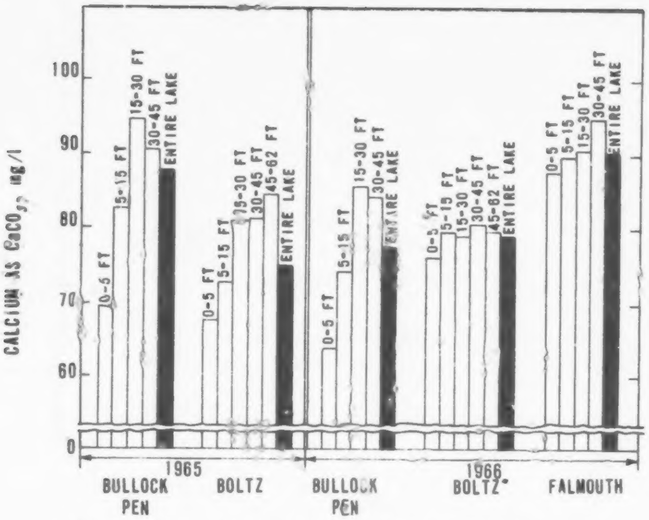

Figure 10. April to October average calcium concentration for each layer and entire volume for each lake.

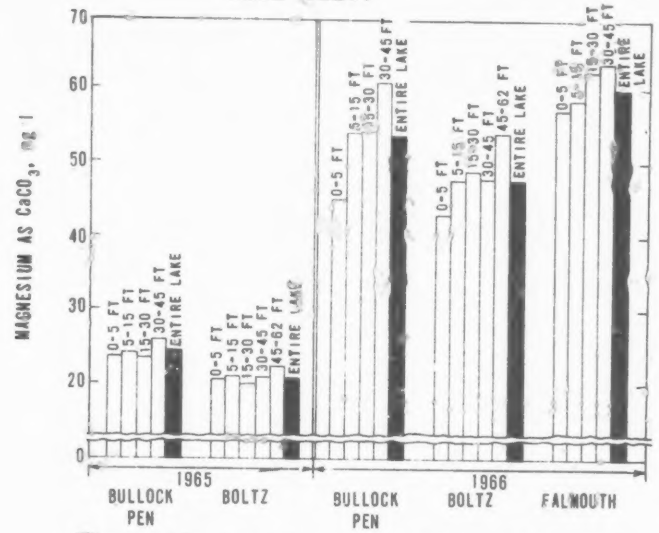

Figure 11. April to October average magnesium concentration for each layer and entire volume for each lake.

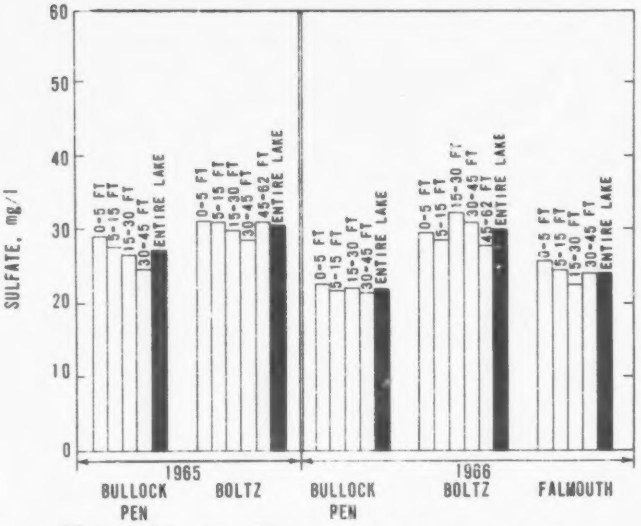

Figure 12. April to October average sulfate concentration for each layer and entire volume for each lake.

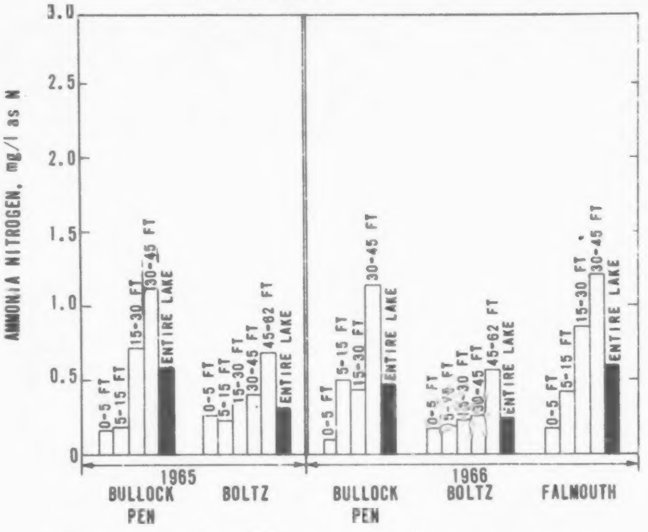

Figure 13. April to October average ammonia nitrogen concentration for each layer and entire volume for each lake.

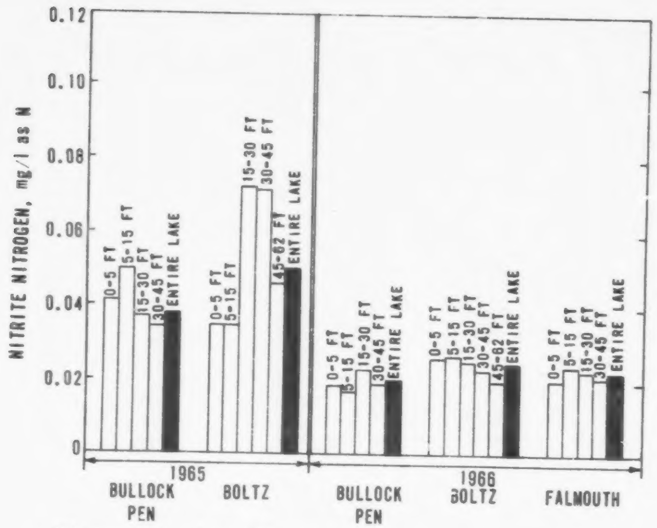

Figure 14. April to October average nitrite nitrogen concentration for each layer and entire volume for each lake.

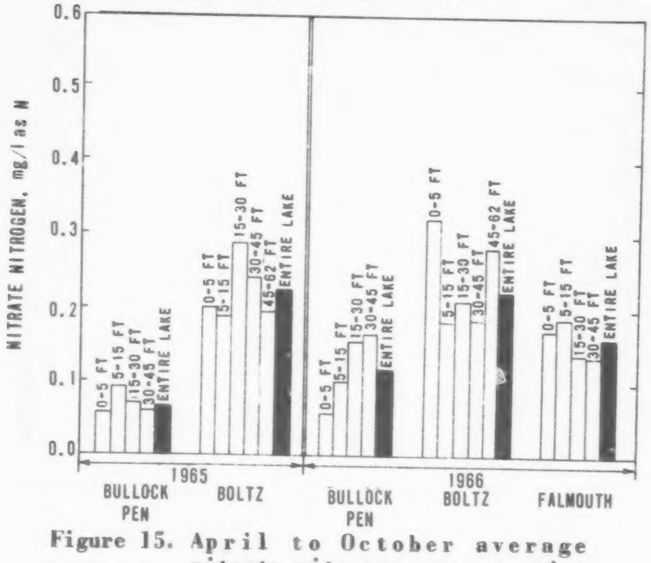

Figure 15. April to October average nitrate nitrogen concentration for each layer and entire volume for each lake.

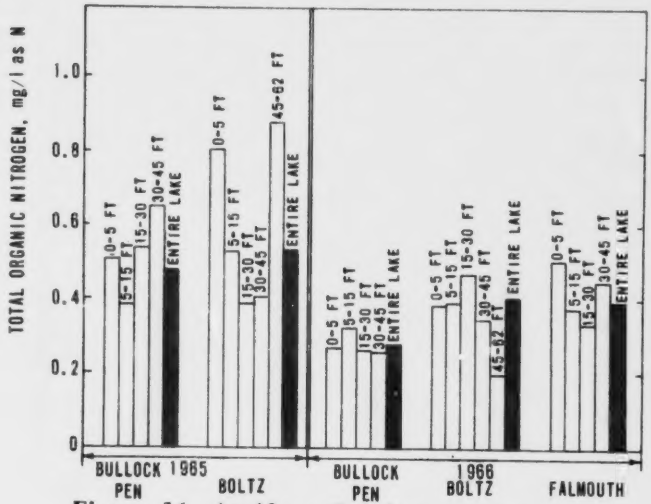

Figure 16. April to October average total organic nitrogen concentration for each layer and entire volume for each lake.

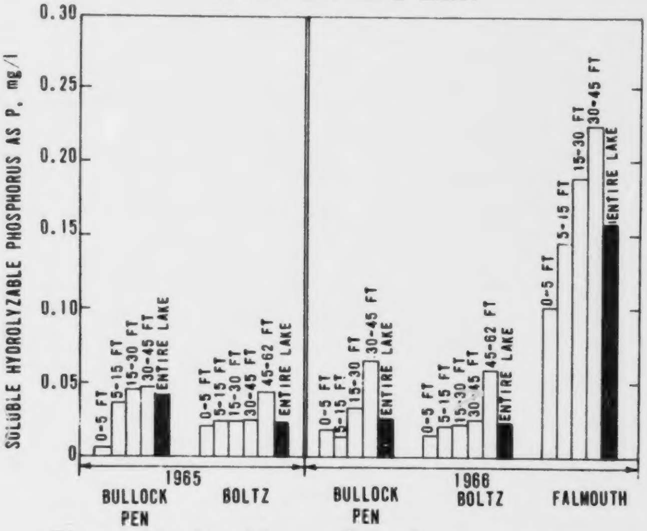

Figure 17. April to October average soluble hydrolyzable phosphorus concentration for each layer and entire volume for each lake.

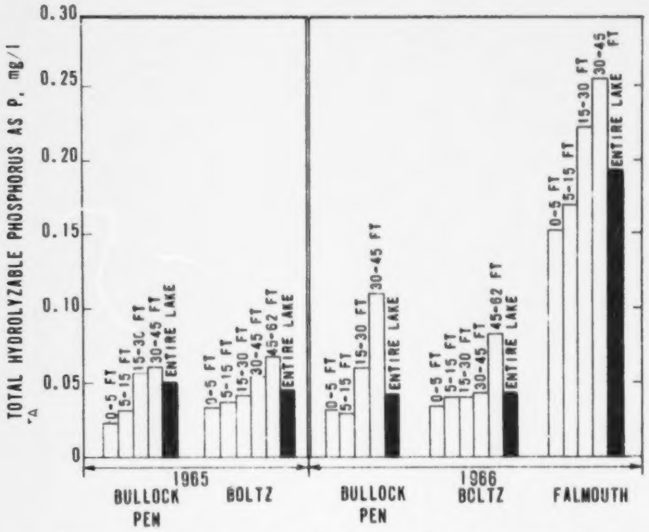

Figure 18. **April to October average total hydrolyzable phosphorus concentration for each layer and entire volume for each lake.**

From June 1 until mid-July all test fish remained alive and apparently normal. In late July, however, specimens in the tank of aerated bottom water died. Specimen-group after specimen-group was killed in 12 to 96 hours of exposure to the aerated bottom water, while all fish in the top water tank remained alive and healthy.

Gill tissues from both groups of fish were treated with hexane; the extracts were checked for toxicity with guppies. The extracts were also examined by gas chromatography using microcoulometric, electron capture, and flame ionization detectors. Studies of the extracts were inconclusive, but stimulated experimentation the following calendar year. In mid-July of 1966, a similar apparatus was assembled at the same site; both top and bottom waters were aerated continuously while in the tank. Again, fish died upon 12- to 36-hour exposure to the aerated bottom water. Tests by emission spectroscopy of gill tissues from the dead fish were negative for heavy metals. Bottom water transferred to the laboratory soon lost its toxicity, although tests were not run immediately. Hexane extracts of fish that were killed and those that survived varied in toxicity when applied to guppies. The gill tissues of dying fish, when examined under a microscope, were not changed noticeably by the toxic material.

Because the tests were inconclusive, the possibility of sulfide toxicity was re-examined. Sulfides were expected to have been oxidized by the presence of at least 4 milligrams of DO per liter of water. The kinetics of sulfide oxidation with DO in natural waters indicates, however, that its oxidation is rather slow, and that it might coexist with DO.

Four stainless-steel tanks were arranged in series, with the original bottom-water tank overflowing into a second, and the second into a third; all were aerated. Test fish were placed in all four tanks and total sulfide analyses were made of the water entering each tank and of the water in each of the four bioassay tanks. Results of sulfide analyses are summarized in Table 2. In spite of the presence of DO, almost 2 milligrams of total sulfide per liter was present in the first tank in the bottom-water series. Repeatedly, test animals died in tank 2, the first in the bottom water series, and survived in the other three tanks. During the second week of November 1966, Bullock Pen Lake overturned naturally, and the toxicity disappeared from the lake.

Although this experimentation indicated the possibility that destratification might bring sulfides to the surface where they might persist for some time before being oxidized by DO, no toxicity to fish was detected during the mixing operation in Boltz and Falmouth Lakes. Apparently, the slow mixing rate prevented sulfide from rising to the surface in a concentration sufficient to be toxic to fish.

Table 2. TOTAL SULFIDES (mg/l) IN TANKS OF AERATED LAKE WATER FROM BULLOCK PEN LAKE, 1966

	Top water		Bottom water					
	Tank 1		Tank 2		Tank 3		Tank 4	
Date	Inflow	Aerated	Inflow	Aerated	Inflow	Aerated	Inflow	Aerated
10/5	0.04	0.01	6.08	2.07	2.07	0.69	0.69	0.15
10/7	0.07	0.03	5.43	1.96	1.96	0.56	0.56	0.12
10/10	-	0.04	-	1.84	1.84	0.53	0.53	0.13
10/11	0.0[illegible]	0.03	6.52	1.83	1.83	0.57	0.57	0.20
10/12	0.04	0.05	6.55	1.87	1.87	0.59	0.59	0.12
10/14	0.05	0.06	6.40	0.75	0.75	0.27	0.27	0.05
10/17	0.05	-	6.47	1.86	1.86	1.14	1.14	0.23
10/19	0.06	0.06	6.18	2.03	2.03	0.71	0.71	0.12

Hydraulic Performance

Although water quality improvement is the primary goal of any artificial destratification operation, whatever mechanical equipment is used to accomplish mixing should be as hydraulically efficient as possible. To compare the hydraulic performance of the mechanical pump and the air diffusion system, the following method was devised.

The time required to create a 2°C temperature change (arbitrarily chosen value) in the deepest portion of the lake was taken as the time required for the mechanical device to circulate the entire contents of the hypolimnion once. This assumption can be made because water withdrawn from the bottom of the lake does not return to the bottom at once, but assumes some intermediate level because of mixing with warm water. Using this technique, the following hydraulic performances were calculated for the pump and the air diffusion system. Of course, the hypolimnion must be circulated more than once to produce uniform temperatures from top to bottom of a given lake.

On August 6, 1965, the pump was used to begin mixing in Boltz Lake. Temperature data taken 15 days later showed that, although 912 hours were required to make the lake isothermal, a temperature rise of 2°C had occurred in the bottom after 360 hours. Based on these data, the pump was moving about 2.7 acre-feet of water per hour. This value is considerably higher than the rated capacity of the pump.

Similar calculations were made with the diffused-air pump system in Boltz and Falmouth Lakes during the first mixing in 1966. In Boltz Lake, with a hypolimnion volume of 1,950 acre-feet, 25 hours were required for one cycle at a pumping rate of about 79 acre-feet per hour. Falmouth Lake, with a hypolimnion volume of 2,050 acre-feet, showed the hypolimnion to be moved once after 25 hours of mixing. This yields a pumping rate of about 82 acre-feet per hour. When these values are compared with the 2.7 acre-feet per hour moved by the mixed-flow pump, the ratio of water moved to energy expended shows a greater efficiency for the air method. This is because the effective diameter of the diffused-air pump was about 100 feet as compared to 1 foot for the mechanical pump.

Mixing the lakes until the temperature was nearly uniform from top to bottom during early June 1966 required 72 hours in Boltz Lake and 134 hours in Falmouth Lake. Temperature data made at various points in the lake from the dam to the upper end showed that in both cases the entire lake was stirred as a unit. As the cold water of the deep area was lifted and warmed, cold, dense water flowed to the deep point because of its greater weight.

DISCUSSION

Radiant energy, precipitation, wind velocity, and wind duration are variables that influence the heat absorption, density profile, runoff, and photosynthesis of reservoirs. Therefore, it must be acknowledged that for comparisons of data from reservoirs, whether stratified, destratified, or prevented from stratifying, exact differences or exact likenesses will not necessarily be produced another year, whether the same or different reservoirs are studied. Trends are more meaningful than exact replications.

Based on the data in Table 1 and Figures 2 through 18, several advantages of artificial destratification can be noted. First, mixing permits more oxygen to be dissolved into the water. Second, although the data show that the total quantity in the lake declined somewhat while mixing, the DO was distributed better at all levels after mixing, and large quantities of organic and inorganic oxygen-demanding materials were eliminated. Third, the presence of DO at all levels decreased the concentration of reduced materials and stopped reducing reactions.

Although reducing conditions may continue to persist in the bottom mud,[12] the presence of DO at the mud-water interface should prevent reductive products from accumulating in the impoundment water. Also, the presence of DO at all levels should enhance the biodegradation of any organic waste materials that might flow into a reservoir.

A more uniform temperature in waters released to the stream should stimulate better growth of fish food organisms, both flora and fauna. This should create better fish growth, both in the impoundment and in the receiving stream by providing a more uniform temperature, greater DO content, more fish food, and greater elimination of materials detrimental to aquatic populations. Circulation of some nutrients to the surface may increase the plankton growth; this also should be beneficial to the fish population.

Decreased surface temperatures should decrease evaporation. Although the evaporation reduction in this experiment was not calculated, a similar experiment conducted by Koberg and Ford[13] showed a reduction of 6 percent in evaporation after artificial destratification.

Two more improvements in general water quality were detected during the artificial destratification experiments. First, autumnal blue-green algal blooms have not occurred in any of the lakes that were mixed. Probably this was because mixing maintains a uniform water quality throughout the water body. This uniformity is conducive to the growth of green algae or diatoms, rather than blue-green algae. Second, a lack of blue-green algal blooms, a lack of anaerobic conditions with their attendant undesirable odors, a more uniform water temperature, and a greater fishing potential should improve the recreational potential of any body of water and afford additional benefits in multipurpose impoundments.

A comparison of operation cost and water circulation rates for mechanical pumping and for air diffuser methods, when reduced to energy input in kilowatt-hours shows an advantage for the diffused-air pump method. Mechanical pumping circulated 0.18 acre-feet of hypolimnion water per kilowatt-hour, so 17 kilowatt-hours would be required to move 1 million gallons of bottom water. The diffused-air pump, on the other hand, moved 3.5 acre-feet of bottom water per kilowatt-hour, so only 0.9 kilowatt-hour would be required to move 1 million gallons of hypolimnion water. At 1 cent per kilowatt-hour, the cost difference is obvious. A compressed air system is easier to install, easier to service, and at this time seems less expensive to operate than mechanical pumping. Moving a large volume of water at low velocity is a more efficient method than moving a small volume of water at high velocity. Although the discharge velocity from the mechanical pump created beneficial currents in the lake so that more water was moved than was actually passed through the pump, the diffused-air system also created beneficial currents. Placement of the diffusers at the deepest portion of the lake, to raise water to the surface caused the remaining cold, dense water in the bottom of the lake to gravitate to the deepest point, with no expenditure of electrical energy.

The effect of rapid mixing on reservoirs that contain large amounts of sulfide is thought provoking. Bullock Pen Lake for 2 years contained enough sulfide, from late August to October, to provide 2 milligrams for each liter of water on the average. This amount in bioassay tanks was toxic to all fish. Would the fish be killed in the lake if the water was mixed rapidly by air diffusers, heavy wind storms, or long-standing low temperatures? It seems entirely possible this might occur. At the beginning of the fourth mixing of Falmouth Lake, however, the water contained, on the average, 3.06 milligrams of sulfide per liter. The lake was mixed for 4 days; no fish died, and the sulfide content fell to 0.22 milligrams per liter throughout the lake. The upper water must have more than a dilution effect on the sulfide.

FUTURE PLANS

Several aspects of artificial destratification still must be investigated. One, the time required to put the entire lake in motion and the benefits to be derived from continuous versus intermittent operation are not known. Such information is important because savings might be effected if impoundments near hydroelectric installations were mixed by periodic use of energy at off-peak hours when electric energy is less expensive. If the total energy requirement were higher for periodic mixing, however, no savings would be realized. Reduced capital costs would result from improved equipment design. For example, other arrangements or groupings of the diffuser stones might increase the rate of water lift. Air bubble size should be studied. Smaller bubbles might improve oxygen transfer from the bubble, as well as lift the water more efficiently.

Two, the influence of destratification on the aquatic organisms in lakes and reservoirs has yet to be determined. Some work is currently (1967) being done on the changes that occur in plankton during mixing, but more detailed studies are needed. Also, little is known about the changes in fish populations and types that occur when a body is destratified.

Three, the toxicity of reaerated bottom water needs additional study. The possible presence of toxic compounds other than sulfides should be investigated. Also, the possibility should be considered that toxic materials or anaerobic conditions will remain at the mud-water interface following destratification, thereby preventing bottom fauna from becoming abundant. Finally, are sulfides or other toxic materials more harmful to one kind of fish than to another, or will certain species be stunted or exist only in small populations in particular lakes because of stratification?

CONCLUSIONS

The following conclusions may be drawn from this study:

1. Artificial destratification can effectively improve water quality in lakes and reservoirs.
2. Mixing provides a nearly uniform water temperature throughout a lake or reservoir; adds DO to the water; decreases the concentration of any reduced materials such as ferrous iron, manganous manganese, and/or sulfides that may be present; stops reducing reactions; and prevents development of large blooms of blue-green algae by providing a uniform water quality.
3. Although the studies are not complete at this time (1967), diffused-air pumping seems a better method for artificial destratification than mechanical pumping.

ACKNOWLEDGMENTS

The authors wish to acknowledge the cooperation of the Kentucky Department of Fish and Wildlife Resources, especially that of Mr. Bernard Carter and his biology staff, who cooperated by granting permission to use the lakes studied and provided morphological data.

Messrs. Lawrence J. Kamphake, George Holtzer, and Richard T. Shibiya, who made the chemical analyses on the lake samples; Mr. Richard E. Thomas, who analyzed the gas content of the diffused air used for mixing; and Mr. Jesse M. Cohen, Dr. Aaron A. Rosen, and Dr. Donald I. Mount, who gave advice and made analyses in connection with the bioassay studies for sulfide toxicity, provided the data upon which this paper was based.

Finally, Messrs. Ernest L. Robinson and Glenn R. Gruber, who worked throughout both seasons operating the equipment in the field and collecting samples, were most helpful in completing this project.

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PAPER NO. 12

IMPOUNDMENT DESTRATIFICATION FOR RAW WATER QUALITY CONTROL USING EITHER MECHANICAL OR DIFFUSED-AIR PUMPING

J.M. Symons, W.H. Irwin, E.L. Robinson, and G.G. Robeck

This paper, the sixth in the series of eight on artificial destratification for reservoir water quality control, discusses the 1966 data on mixing with a diffused-air pump. Attention is given to the effect of mixing on temperature, nitrogen, phosphorus, algal populations, algal types, and the dissolved oxygen budget (including dissolved oxygen, reduced iron, reduced manganese, and sulfides). The influence of multiple mixing with the diffused-air system in 1966 is contrasted with single mixing with the mechanical pump in 1965 for these six parameters. In addition, the usefulness of calculating the "destratification efficiency" (DE) and "oxygenation capacity" (OC), introduced in Papers No. 8 and 9 is further discussed. These two parameters are calculated for this equipment and are compared to other values reported in the literature.

The correct reference citation is:

Symons, J. M., W. H. Irwin, E. L. Robinson, and G. G. Robeck. "Impoundment Destratification for Raw Water Quality Control Using Either Mechanical or Diffused-Air Pumping." Journal American Water Works Association, 59, No. 10, 1268-91. October 1967.

Correction: JAWWA, 60, No. 9, 1069. September 1968.

INTRODUCTION

The potential lowering of water quality in stratified reservoirs and the improvement that occurs after an impoundment is destratified artificially have been discussed in detail in previous papers.[1-4] Although artificial destratification seems to produce no detrimental effects, some questions remain to be answered.

In past studies, impoundments were destratified using a mechanical pump. A 13-acre-feet per day, mixed-flow pump, driven by either a 9- or 15.5-kilowatt gasoline engine was stationed over the deepest part of the lakes to be studied, permitting water drawn from the bottom through a suction pipe to be discharged at the surface. Although improvement in water quality was noted in Vesuvius Lake during the September 1964 mixing and in Boltz Lake during the August-September 1965 mixing, the performance of the mechanical pump, as measured by ''destratification efficiency,''[2,3] was poorer than desired.

The 1964 and 1965 operations were carried out during the late summer and early fall. While water quality was improved, maintenance of good quality water throughout the season seemed to be a better idea than improvement of quality after deterioration. To increase knowledge in this area, two changes were made in the experimental technique. First, mixing was accomplished by a compressor-diffused-air pump system instead of a mechanical pump. Second, mixing was begun during late May and early June, and repeated periodically throughout the 1966 summer. This article compares the performance of a mechanical water pump accomplishing a single mixing in late summer to a diffused-air pump used to mix impoundments periodically throughout the entire season (hereafter called ''multiple mixing''). Comparisons are based on water quality improvement and hydraulic performance.

IMPOUNDMENTS STUDIED

The morphologic characteristics of the lakes studied during the 1965-66 experiments as well as the mixing dates are listed in Table 1. The lake outlines are shown in Figure 1. Bullock Pen Lake was not mixed, and exemplified the water quality characteristics of lakes in northern Kentucky, where the destratification experiments took place. Test Lake 1, Boltz Lake, was artificially destratified by mechanical pumping in 1965 and diffused-air

Table 1. MORPHOLOGIC CHARACTERISTICS OF STUDY LAKES AND DATES OF TESTING

Name	Designation	Volume, acre-ft	Max. depth, ft	Avg. depth, ft	Surface area, acres	Method of mixing	Mixing number	Dates
Bullock Pen Lake	Unmixed	3,200	48	23	142	-	-	-
Boltz Lake	Test lake 1	2,900	62	30	95	Mechanical pumping	1	Aug 6-Sep 10, 1965
						Diffused-air pumping	1 2 3 4	Jun 2-7, 1966 Jun 21-24, 1966 Jul 15-20, 1966 Aug 18-23, 1966
Falmouth Lake	Test lake 2	4,600	42	20	225	Diffused-air pumping	1 2 3 4 5	May 16-19, 20-23, 1966 Jun 10-15, 1966 Jul 1-8, 1966 Jul 26-Aug 3, 1966 Sep 8-13, 1966

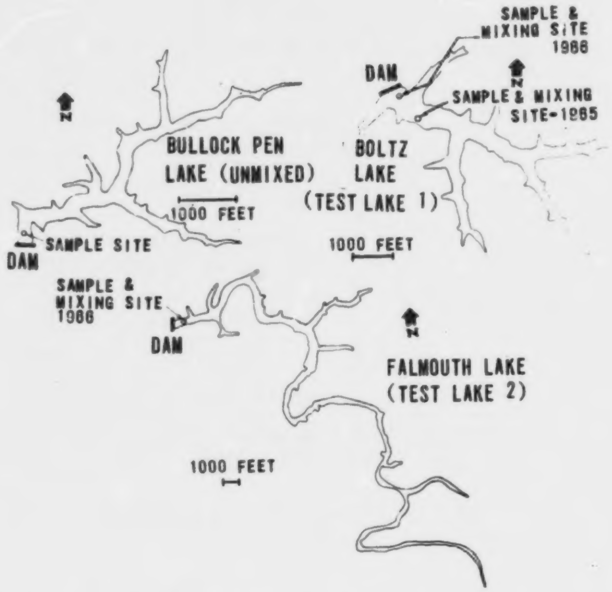
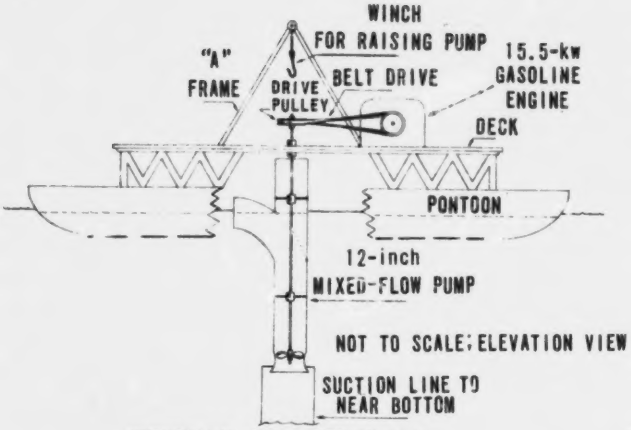

SAMPLE & MIXING SITE 1966
DAM
N
SAMPLE & MIXING SITE-1965
N
BULLOCK PEN LAKE (UNMIXED)
BOLTZ LAKE (TEST LAKE 1)
1000 FEET
SAMPLE SITE
DAM
1000 FEET
SAMPLE & MIXING SITE 1966
DAM
N
FALMOUTH LAKE (TEST LAKE 2)
1000 FEET

Figure 1. Outlines of unmixed and test lakes.

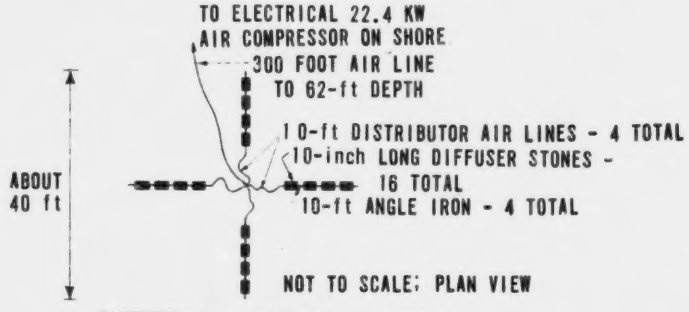
pumping in 1966. Test Lake 2, Falmouth Lake, studied in 1966, was destratified by the diffused-air system only.

EQUIPMENT AND METHODS

A schematic cross section of the mechanical pump used for destratification[1,2] is shown in Figure 2. The dif-


WINCH
FOR RAISING PUMP
"A"
FRAME
DRIVE
PULLEY
BELT DRIVE
15.5-kw
GASOLINE
ENGINE
DECK
PONTOON
12-inch
MIXED-FLOW PUMP
NOT TO SCALE; ELEVATION VIEW
SUCTION LINE TO
NEAR BOTTOM

MECHANICAL PUMP USED IN 1965


TO ELECTRICAL 22.4 KW
AIR COMPRESSOR ON SHORE
300 FOOT AIR LINE
TO 62-ft DEPTH
10-ft DISTRIBUTOR AIR LINES - 4 TOTAL
10-inch LONG DIFFUSER STONES -
16 TOTAL
10-ft ANGLE IRON - 4 TOTAL
ABOUT
40 ft
NOT TO SCALE; PLAN VIEW

DIFFUSED-AIR SYSTEM USED IN 1966
TEST LAKE 1 INSTALLATION

Figure 2. Schematic diagram of the two mixing devices.

fused-air-pump system was a 22.4-kilowatt portable, electrically driven, rotary air compressor attached to a system of diffuser stones by air tubes designed to carry the compressed air to the lake bottom for release. To ensure good distribution of the air to each diffuser stone, an orifice was placed in each hose to create some head loss. Thus, each diffuser received approximately the same volume of air, even if the diffuser apparatus was on an uneven lake bottom. Although resting on the lake bottom, the diffusers did not pick up much mud when in operation. A flow meter placed in the outflow line of the compressor measured the quantity of air supplied to the diffuser system. This system is shown in Figure 2.

The average rate of air released at the 62-foot depth in Test Lake 1 was about 112 cfm, and 115 cfm at the 42-foot depth in Test Lake 2. The compressor was operated on shore, with a plastic pressure hose leading to 16 diffusers arranged in a cross-pattern on the lake bottom. Released air formed a circular pattern, approximately 40 feet in diameter at the lake bottom and 100 feet in diameter at the surface. The aerated water column was less dense than the surrounding unaerated water, and therefore rose a few inches above the natural lake surface. This created a large air-lift pump that moved the bottom water to the surface. The influent hose was weighted and submerged in order to leave the entire lake surface free for recreational use.

To determine the influence of destratification on water quality, samples were taken at various depths near the mixing site in the test lakes, and near the dam in the unmixed lake to test for the following parameters: temperature, concentrations of ammonia nitrogen, nitrate nitrogen, soluble acid hydrolyzable phosphorus,* dissolved oxygen (DO), iron, manganese, sulfide, and phytoplankton count and predominate kinds. Previous work[1] has shown that sampling at any given depth near the deepest part of a lake yields data indicative of the entire lake at that depth.

Details of the analytic procedures used for the physical and chemical determinations have been reported previously.[1,2] To enumerate the plankton, 1-liter samples were collected from various depths in each lake several times during the summer. The samples were preserved with thimerosal, 0.16 percent, plus Lugol's solution, 1 percent. One gram of sodium borate was added for each gram of thimerosal to help keep the thimerosal in solution. This preservative, reported by Williams,[6] aids in settling plankton and preserves all organisms, making identification easier.

The samples were examined in a Sedgewick-Rafter counting cell under 20 X magnification. All recognizable phytoplankters were enumerated and identified to genus. Single-celled organisms and those grouped in colonies were counted as one. Filamentous forms were measured in microns. For example, for *Aphanizomenon*, 100 microns arbitrarily was taken as one unit. Whole zooplankton were enumerated. No special effort was made to identify organisms to species. Diatoms were identified only from wet mounts and no single cells less than 4 microns were enumerated.

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*This has an operational definition and includes all the orthophosphate and polyphosphate and most of the organic phosphate, all converted arithmetically to equivalent phosphorus.

Concentration of some samples became necessary when counts were extremely low. Aliquots of 200- and 500-milliliters, depending upon the need, were filtered through a membrane disc (pore size 0.4 micron). The discs were placed in 250-milliliter beakers and the organisms were scraped gently and rinsed from the disc with distilled water. The slurry was diluted to a particular volume and the concentrated organisms then examined using the Sedgewick-Rafter technique.

RESULTS

Temperature

The upper two parts of Figure 3 show the rising and falling pattern of the water temperature in the unmixed lake at the 5-foot depth during 1965-66. Thermal stratification of lakes in the northern Kentucky area is shown by the low and relatively constant temperature that persisted throughout the summer at 0.7 of the lake's maximum depth. Temperatures at this depth were the same as temperatures at the maximum depth. For the unmixed lake this 0.7 depth is the 30-foot depth. Complete fall circulation is shown at the end of October 1965 when the 5-foot and 30-foot temperatures coincided. In 1966, complete fall circulation occurred in mid-November, as shown by the 2.5°C difference in the upper and lower temperatures at the end of October.

The center portion of Figure 3 shows the increase in temperature at the 0.7 depth (same as bottom temperature) in Test Lake 1 during mixing. At this time, there was a slight decline in surface water temperature. Thermal stratification was largely eliminated in Test Lake 1 during the 5-week mixing period. The lower two portions of Figure 3 contrast the temperature behavior at the 5-foot and 0.7 depth in Test Lakes 1 and 2 during 1966 when diffused-air pumping and multiple-mixing techniques were used. In almost every case, mixing caused a sharp temperature rise in the lower water and some temperature decline in the surface water. Any decline in surface water temperature that occurs during artificial destratification is beneficial because it retards evaporation.

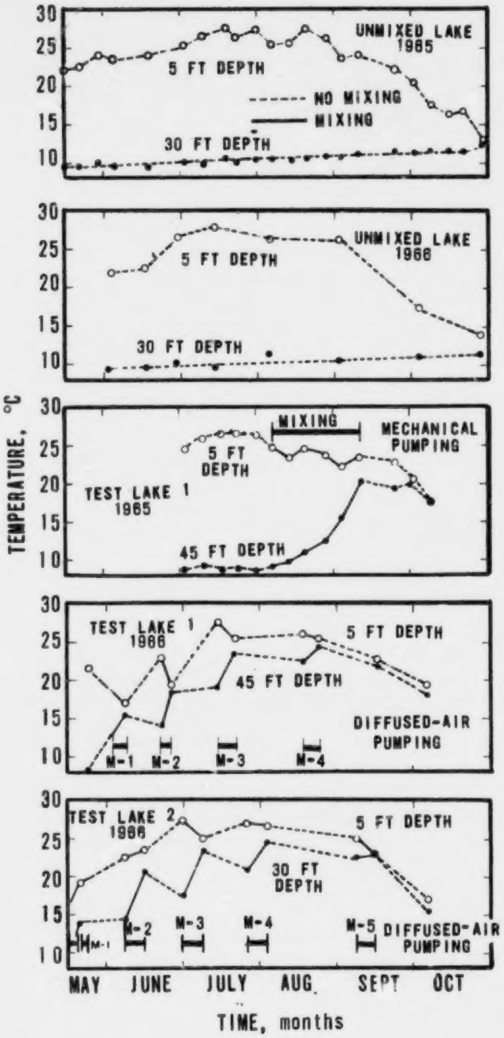

Figure 3. Comparison of temperature at the 5-ft and 0.7 depth in the three study lakes.

Evidence of restratification between mixings is seen in the lower two portions of Figure 3. Usually the 5-foot depth temperature increased, but in Test Lake 2 some cooling occurred at the 0.7 depth, possibly because of loss of heat to the mud bottom.

Nitrogen and Phosphorus

The upper portion of Figure 4 shows that the sum of the ammonia nitrogen concentrations at the 5-foot depth in the unmixed lake remained at about 0.2 to 0.3 milligram per liter throughout the 1965-66 studies. Similar data for Test Lakes 1 and 2 show that artificial destratification by both techniques increased the concentration of soluble nitrogen near the lake surfaces in some cases, but later data indicated that this increase did not stimulate algal growth.

The upper portion of Figure 5 shows that during 1965-66 the soluble phosphorus concentration at the 5-foot depth in the unmixed lake usually varied between 20 and 50

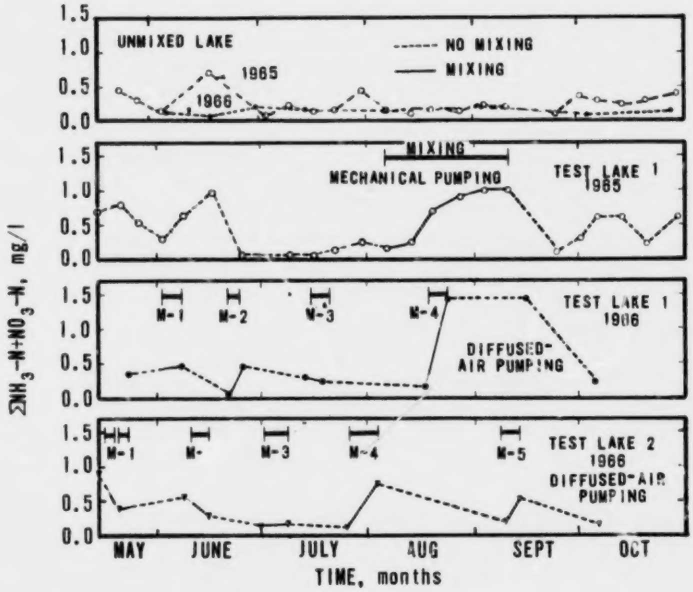

Figure 4. Sum of NH_3-N and NO_3-N concentrations at the 5-ft depth in the three study lakes.

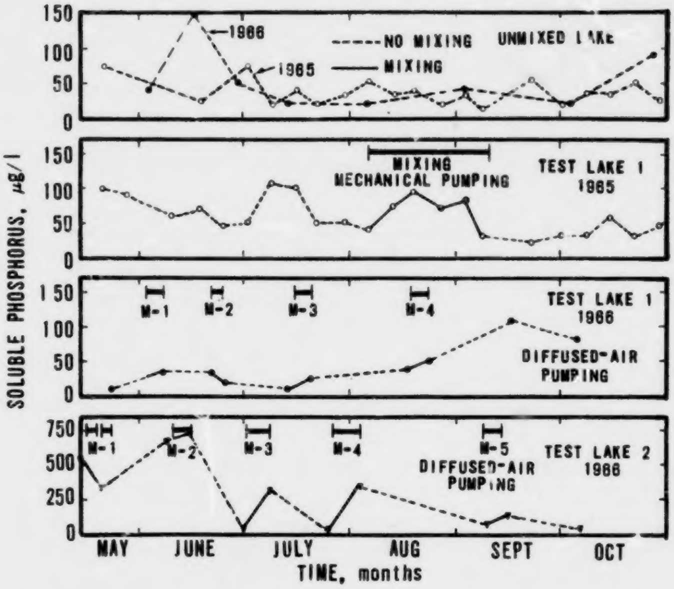

Figure 5. Soluble phosphorus concentrations at the 5-ft depth in the three study lakes.

micrograms per liter, although concentrations were higher at times. Soluble phosphorus concentrations at the 5-foot depth in Test Lakes 1 and 2 increased somewhat during artificial destratification. Test Lake 2 was fertilized heavily with commercial fertilizer during 1966, in an attempt to improve fish production. Therefore, the soluble phosphorus concentration in this lake was considerably higher than in Test Lake 1. Note the scale change in the lower portion of Figure 5 as compared to the center and upper portions. The rise in concentration of soluble phosphorus during the third mixing in Test Lake 2 is nearly 300 micrograms per liter. As in the case of nitrogen, however, despite the increase in soluble phosphorus, algal blooms did not occur.

Plankton Concentrations

During 1965 phytoplankton populations (top portion Figure 6) rose in the early summer near the surface of the unmixed lake. A concentration of approximately 2,000 to 3,000 organisms per milliliter was maintained throughout the summer. Similar data in 1966 were more erratic, the counts being lower than in 1965. The middle portion of Figure 6 shows that at the 5-foot depth in Test Lake 1 the counts declined during all the mixing periods except one, despite some increase in nitrogen and phosphorus in the surface during artificial destratification. The reason for the decline in phytoplankton is not yet (1967) known.

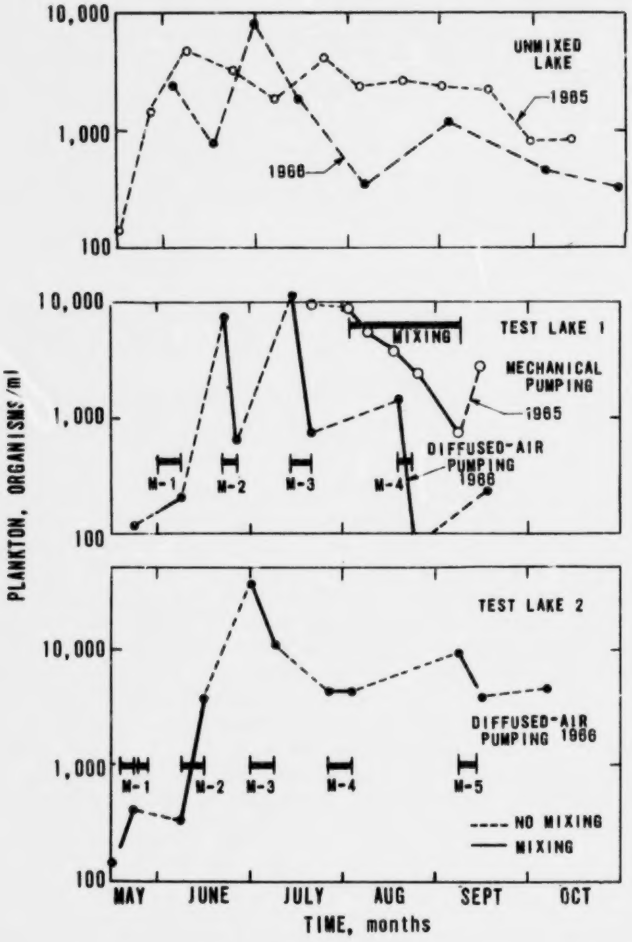

Figure 6. Plankton counts at the 5-ft depth in the three study lakes.

The rapidly changing altered environment may have been unfavorable to the algal population. Previous studies[2] indicated that surface clarity was not impaired during mixing. Whatever the reason, the effect of mixing was only temporary, because after mixing stopped the phytoplankton populations rose again.

Similar declines in surface algal counts occurred during mixing 3 and mixing 5 in Test Lake 2 (bottom portion Figure 6). The high count before the start of mixing 3 occurred because filamentous organisms were predominant, and 100 microns of length was chosen arbitrarily to be reported as one organism. During mixing 4 the plankton count remained nearly constant.

Although there was some rise in plankton population near the surface during the spring and in some cases after mixings stopped, the decline in surface population that occurred during mid and late summer mixings and the subsequent rise to levels equal to or less than previous populations show that the plant nutrient increase that occurred during artificial destratification did not cause algal blooms.

Standing Crop

Although the data in Figure 6 show that the phytoplankton concentration in the upper zones of Test Lakes 1 and 2 declined during most mixings, they do not indicate whether the standing crop was reduced during artificial destratification or whether the plankters that were near the surface before mixing were simply mechanically moved to lower depths during mixing. Figure 7 shows that a reduction in standing crop actually did occur during most mixing operations, and that destratification did not merely transport surface algae to greater depths in the lakes. The data in Figure 7 were obtained by converting the plankton count at each depth to total numbers of algae by multiplying the counts per unit volume by the total water volume in a given layer. An arithmetic summation of these data then yielded the approximate total number of organisms in the lake. The right-hand scale shows the average phytoplankton concentration throughout the lake. As with the 5-foot-depth plankton count, whatever environmental change caused a decline in the standing crop during mixing was overcome by the algal populations after it stopped, as shown by the rise in standing crop between most mixing periods.

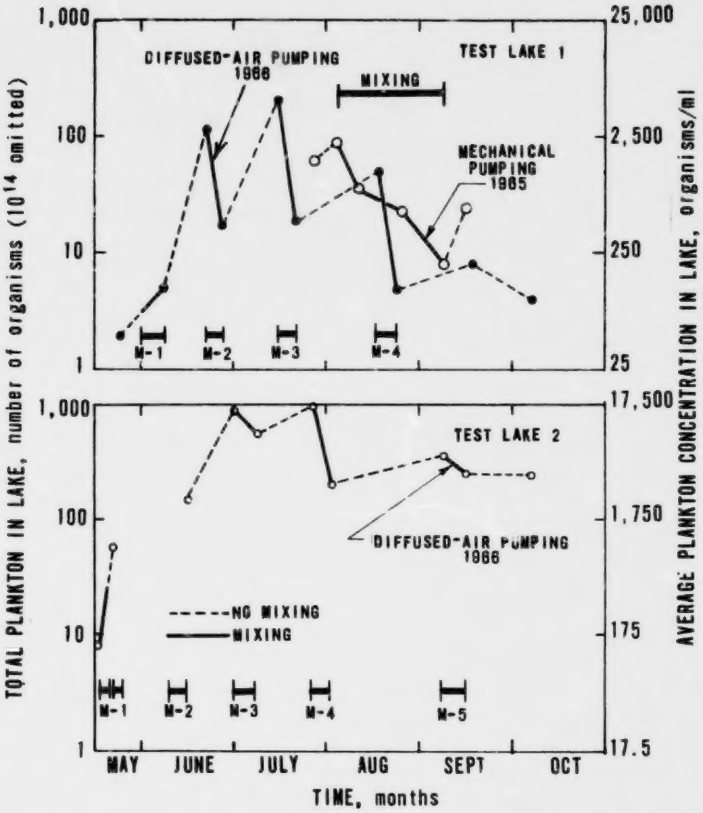

Figure 7. Total plankton numbers and average plankton concentrations in Test Lakes 1 and 2.

Change in Algal Type

There have been indications[1,2,5] that artificial destratification tends to alter the environment in impounded waters to inhibit the growth of blue-green algae so as to change the predominant algal populations from blue-green to green. Figures 8 to 11 present data from these experiments that confirm past conclusions. Figure 8 shows that during 1965-66 the green and blue-green algal counts at the 5-foot depth in the unmixed lake were fairly steady throughout midsummer, and were equal to one another, approximately at 500 to 1,000 organisms per milliliter. Figure 9 shows that in Test Lake 1 during the 1965 period

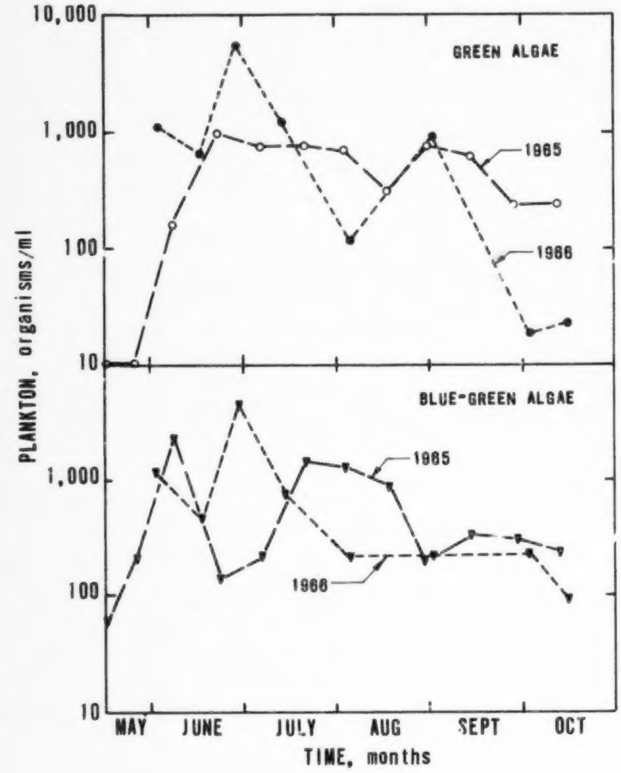

Figure 8. Green and blue-green algae counts at the 5-ft depth in the unmixed lake.

of artificial destratification, the count of both types of algae declined, with blue-green counts declining slightly more than green. In this same lake in 1966, however, a much greater drop occurred in the blue-green algal populations than in the green during all mixing operations (Figure 10).

Figure 11 shows similar results for Test Lake 2. During mixing 3 and mixing 4 the environment was changed sufficiently to cause a slight increase in the green algal populations while the blue-green populations were declining. As noted previously, a large population of filamentous *Oscillatoria* was present at the beginning of mixing 3; therefore the blue-green algal count was very high.

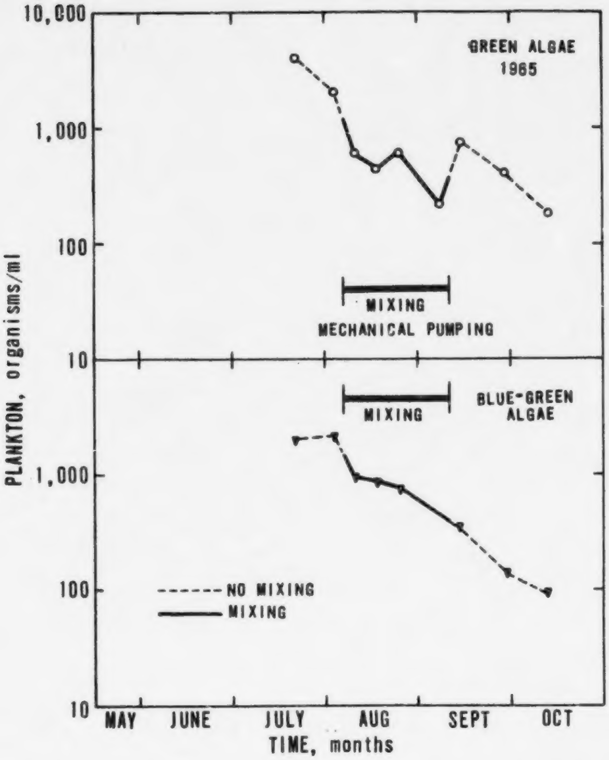

Figure 9. Green and blue-green algae counts at the 5-ft depth in Test Lake 1.

The conclusions reached from counting and identifying the algae during mixing in the two test lakes are in agreement with those stated by Wirth.[7] To quote:

"After mixing was initiated in 1966, water clarity improved as shown by the average transparency values of 1963 and 1965. Although an initial bloom of algae might have been expected due to pumping nutrients into the epilimnion when mixing began, such did not occur. In fact, transparency improved during the first two weeks of operation until a bloom began during the 3rd week in July, which diminished by August. From August into September, water transparency continued to gradually improve, but decreased to about 6 feet in October. At no time during

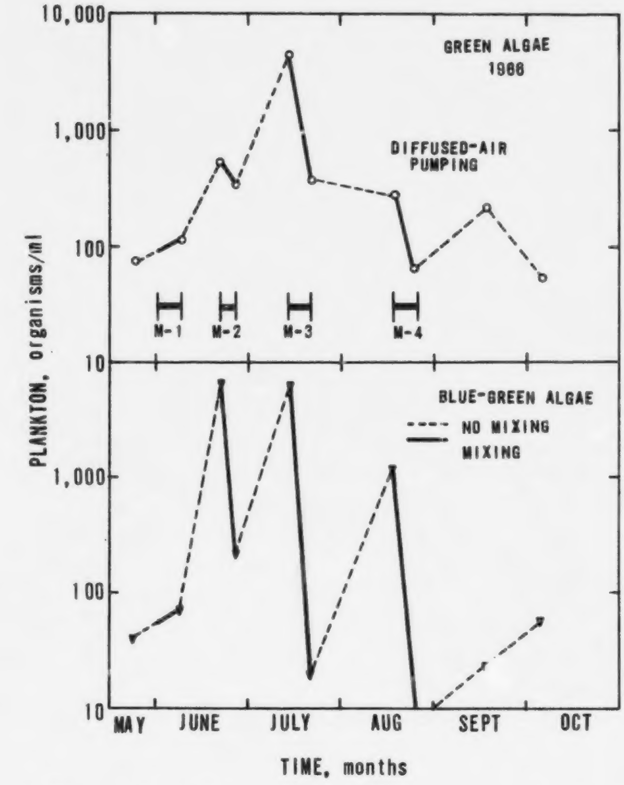

Figure 10. Green and blue-green algae counts at the 5-ft depth in Test Lake 1.

the summer and fall period did an offensive nuisance algae bloom occur such as were noted in previous years.''

DISCUSSION

Dissolved Oxygen Balance

It has been shown[2-4] that supplying DO for reaeration and oxidation of reduced substances in lake waters is one major water quality improvement that occurs during artificial destratification. Comparison of the oxygen transfer capacities of the two destratification devices used in 1965 and 1966 is difficult, because of the varying

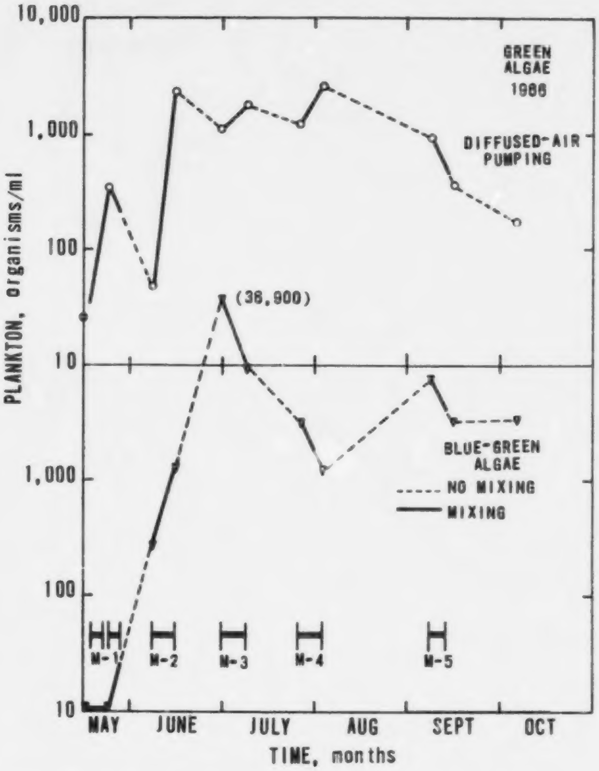

Figure 11. Green and blue-green algae counts at the 5-ft depth in Test Lake 2.

depths of the three study lakes. The thermocline depth of lakes in the same geographic area is usually similar, however, irrespective of total depth. Thus, the zone of zero DO water frequently begins at a similar depth in all lakes in the same locale. Inspection of the data taken during this study shows that, with respect to the concentrations of DO, sulfide, iron, and manganese, the water below the 30-foot depth behaved similarly in all three lakes. Zero DO water below the 30-foot depth is reasonable, as similar results were obtained in studies by Walesh[8] on a lake of similar depth in the Baltimore area and from unpublished results of the U.S. Corps of Engineers from much deeper bodies of water at similar latitudes.

To determine the ''free'' DO surplus or oxygen demand (DO deficit) in the bottom portion of the three lakes in this study, the concentrations of DO, sulfide as oxygen equivalent (2 times the concentration), manganese as oxygen equivalent (0.29 times the concentration), and iron as oxygen equivalent (0.29 times the concentration) below the 30-foot depth were multiplied by the water volume to calculate the number of pounds of each constituent. Inspection of COD data showed that changes were too slight to be included in the DO balance. The algebraic sum (plus for DO and minus for oxygen sinks, sulfide, iron, and manganese) yielded the oxygen surplus or deficit in these lower waters. The net oxygen surplus or deficit calculated was divided by the volume below the 30-foot depth to adjust the data for the varying volumes of the three lakes under study. For these calculations, all of the iron and manganese measured in the samples was taken as the reduced form.

The DO balance data for the unmixed lake in Figure 12 shows that ''free'' DO was eliminated below the 30-foot depth about July 1 during 1965-66, and that sulfides and reduced iron and manganese continued to accumulate throughout the remainder of the summer and fall, as indicated by increasing negative values. Fall overturn, which occurred late in October 1965 and early in November 1966, brought these reduced substances into the oxygenated zone and eliminated them prior to winter.

Oxygenation Capacity

Before mixing began in early August 1965 in Test Lake 1, the downward trend of DO balance data had already occurred (Figure 13). After mixing started on August 6, 1965, reducing conditions were strong enough and reaeration was slow enough so that the oxygen sinks continued to increase in quantity during the first week of mixing. Thereafter, oxygen was introduced into this layer and the oxygen sinks were rapidly oxidized. After mixing stopped, reduced substances again began to form, but soon thereafter mixing occurred by wind and descending cooler water from the surface, and ''free'' DO was once again found below the 30-foot depth.

To evaluate the ability of any destratification device to add DO and oxidize oxygen sinks ''oxygenation capacity'' (OC) has been defined as follows:

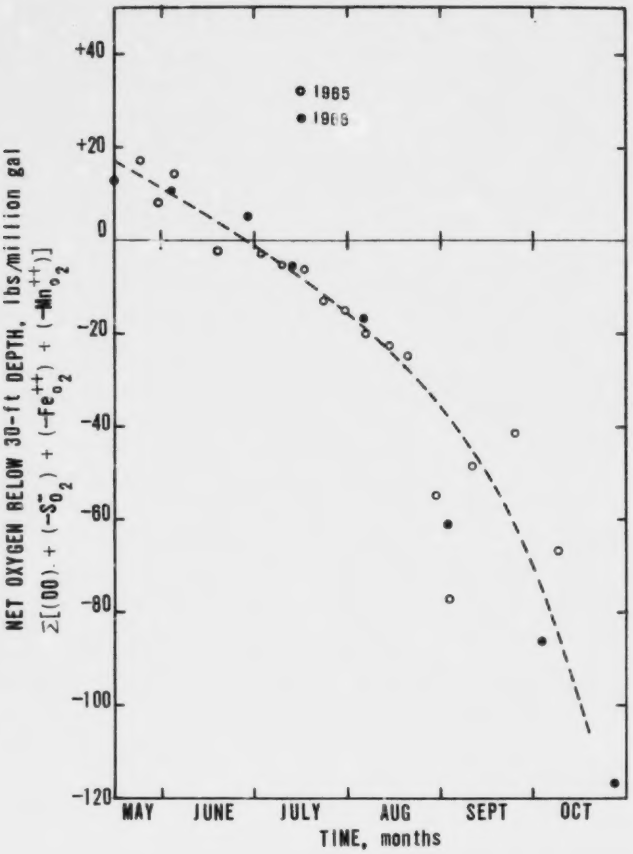

Figure 12. Oxygen surplus or deficit below 30-ft depth in unmixed lake.

$$\text{Oxygenation Capacity} = \frac{\text{Net change in oxygen balance from } t_1 \text{ to } t_2}{\text{Total energy input from } t_1 \text{ to } t_2} \quad (1)$$

in which t_1 is the time of start of mixing, and t_2 is the time of end of mixing. The units of OC are pounds of DO transferred per kilowatt-hour.

To calculate the OC of the mechanical pump, the rate of oxygen sink production was extended (see dotted line) to

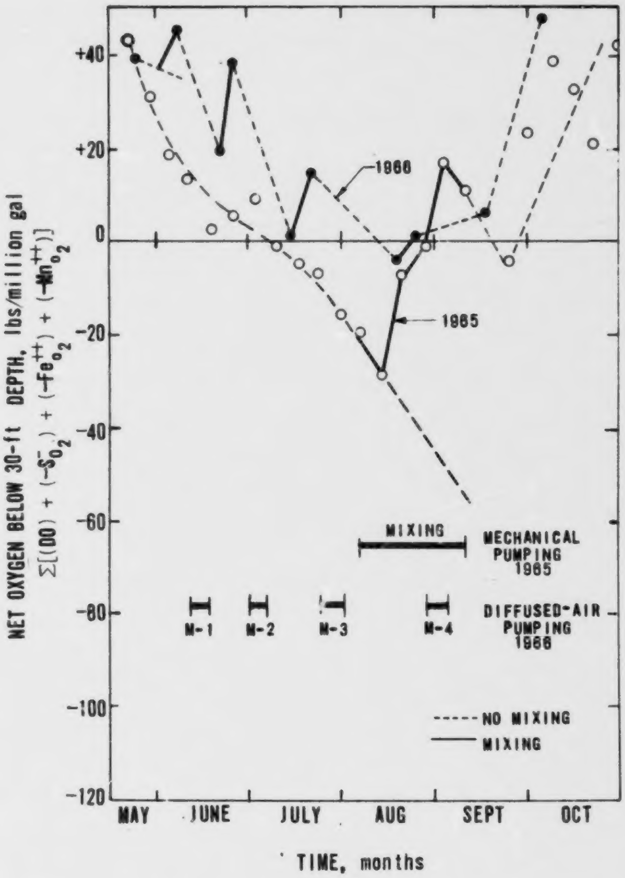

Figure 13. Oxygen surplus or deficit below 30-ft depth in Test Lake 1.

the end of the mixing period. At this time, the difference between the actual condition of the lake water and the condition that would have prevailed had there been no mixing was taken as the amount of oxygen transferred to the water by the mixing device. In this case, 66 pounds DO per million gallons was transferred to the water. Multiplying by the total water volume below the 30-foot depth and dividing by the total energy input yields an OC of 1.3 pounds DO per kilowatt-hour. This value differs slightly from the OC reported earlier for this device, (1 pound DO per kilowatt-hour)[3] because a different method of calculation was used. This difference is unimportant, however.

Single and Multiple Mixing

The difference between single mixing in 1965 and multiple mixing in 1966 in Test Lake 1 is also shown in Figure 13. Multiple mixing of this lake during 1966 almost completely prevented the formation of reduced substances in the volume of water below the 30-foot depth. In an attempt to compare the OC of the diffused-air pumping system and the mechanical pumping system, *net* changes in DO balance (corrected by extending the trend of the data between any two mixings through the mixing period in question) were calculated per unit of energy input. Because no data were taken just prior to mixing 1, the trend of the unmixed lake was projected from the first datum point through mixing 1 (see dotted line). The OC of the diffused-air pump for each of the four mixings and the OC of the mechanical pump is shown in Table 2.

Table 2. OXYGENATION CAPACITY OF BOTH DESTRATIFICATION DEVICES

Test lake	Mixing No.	Year	Method	Oxygenation capacity,[a] lb DO kw-hr
1	1	1965	Mechanical pumping	1.3
	1		Mechanical pumping	0.7[b]
1	1	1966	Diffused-air pumping	1.6
	2		Diffused-air pumping	4.3
	3		Diffused-air pumping	3.3
	4		Diffused-air pumping	1.0(?)
	1-4		Diffused-air pumping	1.7[b]
2	1	1966	Diffused-air pumping	0.0
	2		Diffused-air pumping	1.4
	3		Diffused-air pumping	3.0
	4		Diffused-air pumping	4.2
	5		Diffused-air pumping	3.9
	1-5		Diffused-air pumping	0.02[b]

[a] These data can be used as a guide only, as they are strongly influenced by conditions in the water at start of the test (Figure 15).

[b] Calculated by the area method, see text.

Because Test Lake 2 was heavily fertilized to stimulate fish production, the intensity of reducing reactions was much greater than in Test Lake 1 (Figure 14). Here,

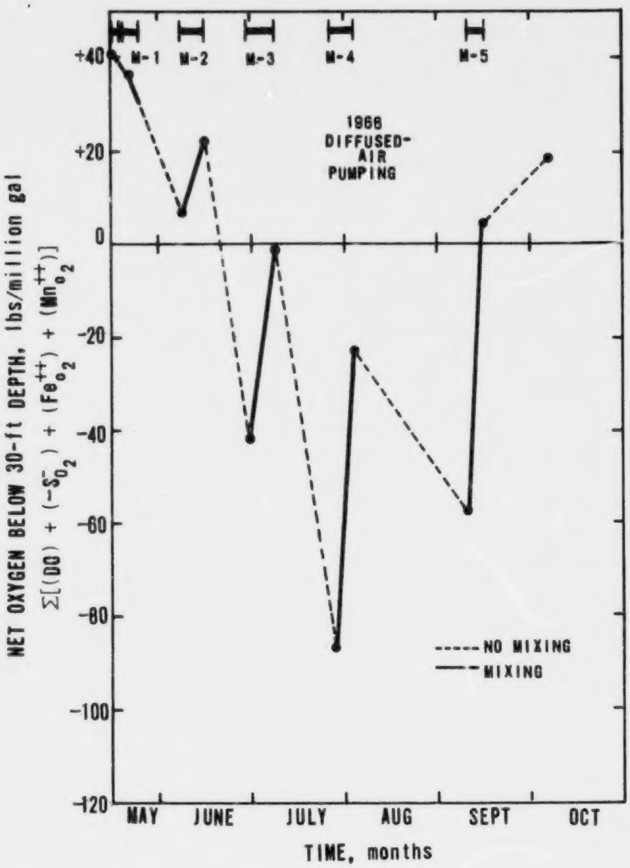

Figure 14. Oxygen surplus or deficit below 30-ft depth in Test Lake 2.

despite multiple mixing by diffused-air pumping from late June to mid-September 1966, no ''free'' DO existed below the 30-foot depth. As shown, however, each mixing did oxidize considerable quantities of reduced material in this water zone. This was true especially during mixings 3, 4, and 5. The OC of the diffused-air pump in this lake is also summarized in Table 2.

During restudy of these data subsequent to publication, an alternate method of calculating an OC for a given piece of mechanical equipment was developed. This technique involves determining, with a planimeter, the area between the control and test DO balance curves during the entire study period; for example, August 6 to September 10 for Test Lake 1 in 1965, and June 2 to August 23 in 1966. The units of this area, after multiplying by the volume below 30 feet in the test lake, are ''pounds DO-hours.'' The OC is then calculated by dividing this area by the area under the cumulative energy input curve during the same time period. This area has the units ''kilowatt-(hours)^2''.

The resulting ratio gives an OC value that may be more indicative of the performance of a destratification device over an entire season, and therefore may be a better design parameter than an OC developed over a short period. Values for the OC calculated by the ''area method'' during the 1965 and 1966 tests are included in Table 2 for comparison. Further discussion of this parameter for design purposes is contained in Paper No. 14 of this Compilation. The authors are indebted to Mr. J. Keith Carswell, Research Engineer, Bureau of Water Hygiene, for making these calculations.

The OC data given in Table 2 for the area method produce lower values than those for the previous method of calculation. The extremely low value for Test Lake 2 results from comparing the behavior of this lake to an improper control. The heavy fertilization of Test Lake 2 made it much more active with respect to DO depletion than Bullock Pen Lake, the control. Therefore, a previous year of record for Test Lake 2, if available, would have been a better control. As with any method of performance evaluation, choice of a control is important for calculation of an OC value. Only through further study and use of the area method for calculation of OC will its true worth be revealed.

Oxygen Transfer Efficiency

The data in Table 2 are somewhat confusing, unless each value is related to the concentration of DO or oxygen sinks in the lower waters of the lake at the start of each mixing. These data are shown in Figure 15. Here, the resulting OC for each mixing is plotted against the DO balance (plus if DO is present, minus if reduced materials are present) at the start of each mixing. If the water were saturated with DO at the start of a test (about 70 pounds DO per million gallons), no more oxygen could be added and the OC would be zero. As the DO concentration

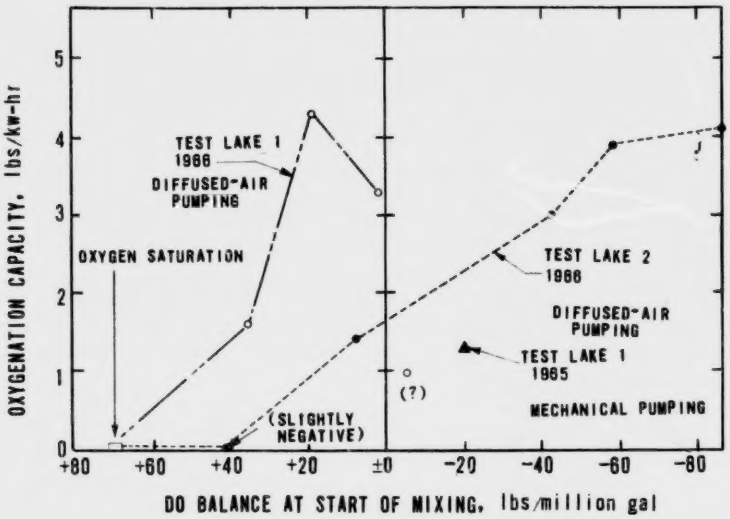

Figure 15. Relationship of oxygenation capacity to condition of DO balance in lake at start of destratification.

declines, oxygen transfer becomes more efficient and the OC rises. When DO is absent and high concentrations of reduced materials exist at the start of a test (high negative DO balance), their oxidation is rapid and a high OC results. The longer a destratification device operates the lower the OC will be, because oxygen transfer per unit energy input becomes increasingly lower as DO approaches saturation.

Figure 15 shows that the OC of the mechanical pump was much lower than that of the diffused-air pump on an equivalent basis. This is probably because the diffused-air pump was more efficient than the mechanical pump in moving water to the surface for reaeration, not because of oxygen transfer from the diffused-air bubbles to the water, although some of the latter must have occurred. An attempt to determine oxygen loss in the rising bubbles was made, but nitrogen and oxygen content were about the same as in air, indicating little oxygen transfer from the bubbles.

Using 0.016 pounds of oxygen per cubic foot of air, about 2,600 pounds of oxygen per day is discharged from the diffusors at an air flow of 115 cfm. For example comparison, during mixing 3 in Test Lake 2 (see Figure 14) the rate of oxygenation was about 750 pounds of oxygen per day, only about one-fourth the amount of oxygen delivered by the compressed air. Only an accurate analysis of the off-gases will reveal what percentage of the reaeration is from transfer from the bubble and what percentage is from atmospheric reaeration. Bernhardt[9] found that 75 percent of the added oxygen was lost, based on analysis of the rising bubbles with an Orsat apparatus. If bubble reaeration is significant, diffused-air pumps might have an advantage over mechanical pumps for some applications.

Table 3 summarizes the OC of other artificial destratification and reaeration devices. The data in Table 3 must be viewed as a guide, because they are influenced by testing conditions.

Table 3. OXYGENATION CAPACITY OF OTHER ARTIFICIAL DESTRATIFICATION AND REAERATION DEVICES

Type and reference	Oxygenation capacity,[a] lb DO/kw-hr
Artificial destratification	
Unconfined diffused air[9]	1.2
Unconfined diffused air[9]	0.3
Confined diffused air[9]	4.3
Aero-Hydraulics Gun[9]	1.4
Reaeration	
Turbine[10]	2.7-4.0
Turbine[11]	5.0
Turbine[12]	3.4
Floating turbine[13]	2.0-2.4-low river flow 5.4-6.0-high river flow
D - O Aerator[14]	3.3
Diffused-air activated sludge[15]	3.1-3.3

[a] These values may be used as a guide, but are influenced by test conditions.

Destratification Efficiency

In spite of the limitations,[3] calculation and comparison of the "destratification efficiency" (DE) is a useful way of comparing the mechanical performance of artificial destratification devices. The DE for any lake mixing operation is calculated below:

$$\text{Destratification Efficiency (DE)} = \left[\left(\frac{\text{Net Change of Stability from } t_1 \text{ to } t_2}{\text{Total Energy Input from } t_1 \text{ to } t_2}\right) \times 100\right] \quad (2)$$

in which t_1 is the time of start of mixing and t_2 is the time of end of mixing. An example calculation of DE is available from the authors.[16] The stability trend in lakes in the northern Kentucky area, rising to a peak in mid-July and then declining toward an autumn overturn, is shown in Figure 16.

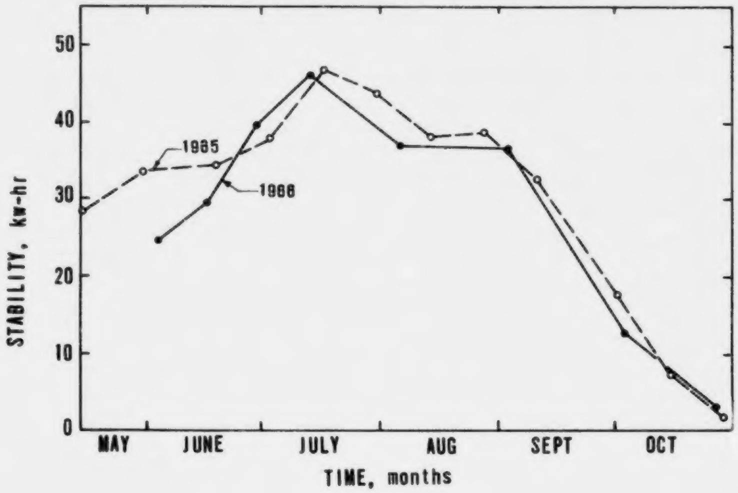

Figure 16. Stability of the unmixed lake during 1965 and 1966.

One difficulty in assessing the mechanical performance of an artificial destratification device by following the stability trend during mixing is determining how the stability pattern observed during destratification deviates from the trend that would have occurred had no destratification taken place. An experiment of this type should have a control. But in this case, because no two lakes are identical, exact controls for destratification experiments are impossible. Two techniques are available for approximating the nonmixed stability behavior in a test lake during mixing, so that the net change is available to use in the numerator of the fraction in Equation (2).

COMPARING STABILITY PATTERNS

One technique, applicable to the single mechanical pumping of Test Lake 1 in 1965, is first to show that the stability pattern in both the test lake and another lake nearby are relatively similar when no mixing is occurring. From May to August 1965, the stability patterns in the unmixed lake and Test Lake 1 were similar, although at any given time the stability in Test Lake 1 was higher because of the difference in morphologic characteristics. Then, an approximate control may be obtained just prior to artificial destratification by adjusting the stability of the nearby lake to match the stability of the test lake.

This has been done in Figure 17, in which the 1965 data from Figure 16 have been adjusted to match the Test Lake 1 data in mid-July. Using this technique, the trend of the Test Lake 1 stability during artificial destratification may be compared to what might have been the trend of the stability in the lake had it not been mixed. At the end of the mixing period, therefore, the *net* change in stability that occurred because of the mixing may be determined.

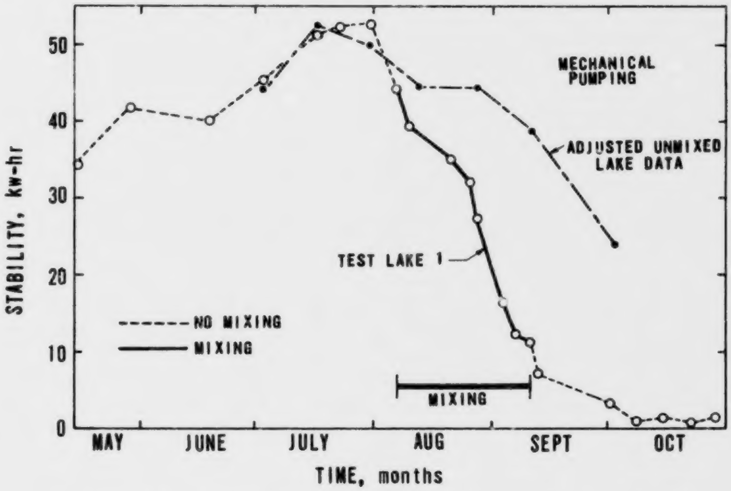

Figure 17. Stability of Test Lake 1 during 1965.

The technique of using the stability pattern of an unmixed lake as a "control" can only be employed during the first mixing of a multiple-mixing experiment, such as occurred in 1966. After the first mixing, the temperature in a test lake is so altered that its behavior between mixings does not follow the pattern of any other body of water. In Figure 18, adjusted, unmixed lake data

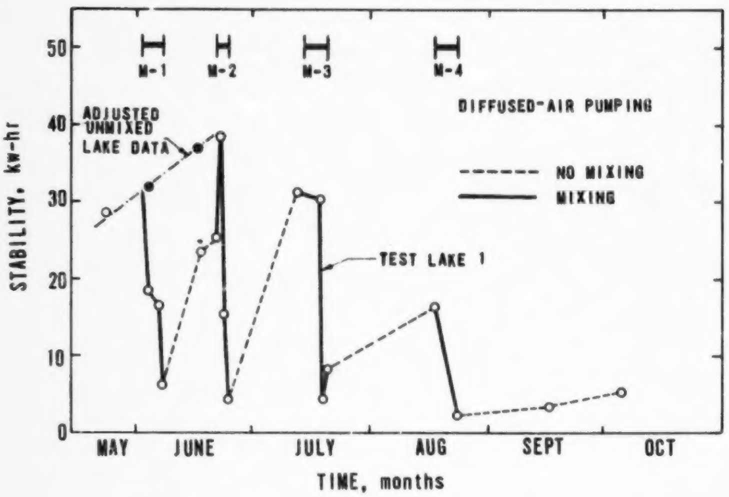

Figure 18. Stability of Test Lake 1 during 1966.

were used to obtain the net change in stability caused by the diffused-air pump during only the first mixing in 1966.

Thereafter, the second technique that may be used for obtaining net stability changes during mixing was employed. Here, the trend of the stability data in the lake between mixing periods was extrapolated through the mixing period to determine what it would have been at the end of the mixing period had no mixing taken place. These data would have been improved had several data points been available just prior to mixing, rather than just assuming that the trend from the last datum point of the previous mixing and the datum point taken just before a given mixing established the trend of stability between mixings. Figure 18 shows that each mixing markedly reduced the stability, and a subsequent rise in stability occurred between each mixing. The rate of stability rise between mixings was most rapid during midsummer and, because of generally cooler air temperatures, was less rapid between mixings 3 and 4.

Similar results are shown for Test Lake 2 in Figure 19. Here, as before, adjusted unmixed lake data were used only during the first mixing period in May, but thereafter the stability data taken between mixings were extrapolated to determine the net influence of mixings 2, 3, 4, and 5

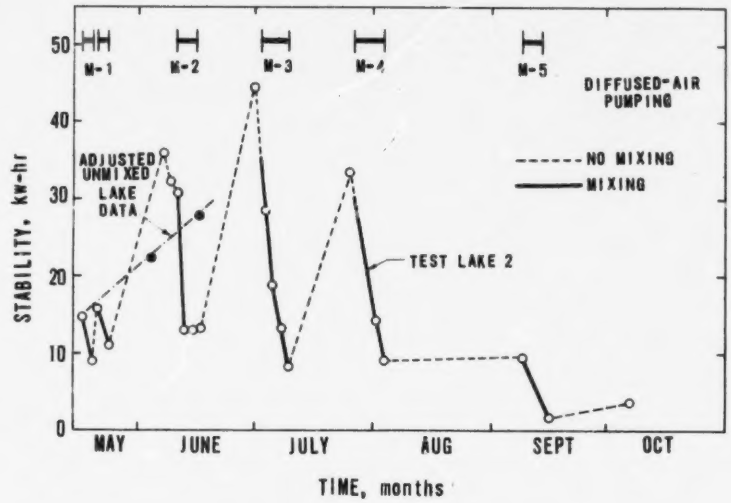

Figure 19. Stability of Test Lake 2 during 1966.

on the stability. As in Figure 18, the rate of stability rise between mixings declined as the summer progressed.

AREA METHOD

In a manner similar to that described in the section of this paper entitled ''Single and Multiple Mixing,'' the DE for a mixing device may be calculated for an entire season by dividing the area between the test and control stability curves by the area under the cumulative energy input curve. The advantages and disadvantages of this technique, noted in the previous section, also apply to the DE calculation. The use of DE as a design parameter is discussed further in Paper No. 14 of this Compilation. DE values for the 1965 and 1966 experiments are included in Table 4 for comparison purposes. The authors are indebted to Mr. J. Keith Carswell, Research Engineer, Bureau of Water Hygiene, for making these calculations.

Table 4 shows the DE for the two types of mixing devices used in these experiments and for the various mixings during 1966. For the 1966 data, in addition to calculating a DE for each mixing, the individual energy inputs and declines in stability were summed and an overall DE calculated. These data show that, in general, the diffused-air pump had a better hydraulic performance than the mechanical pump used in 1965. The DE calculated for the mechanical pump may not be applicable to all mechanical pumps, however, because the pump used in this experiment generated unnecessarily high headlosses.

During 1966, the diffused-air pump had a higher calculated DE in the deeper, smaller surface area lake, Test Lake 1, than in the shallower, larger surface area lake, Test Lake 2. In Test Lake 2, solar radiation on the large portion of the lake volume that was near the surface tended to reform stable conditions very easily, even during mixing. Therefore, more energy will be required to reduce a given stability towards zero in a lake whose surface area is relatively large in proportion to depth.

FACTORS INFLUENCING DE

Table 4 also summarizes data from several other destratification operations. These data must be viewed with caution. In many cases, the change in stability used in the numerator of Equation (2) was uncorrected for any natural changes that might have occurred during mixing. The data shown in the last two columns must be viewed as a guide only, because of the many factors influencing these values.

Here are several examples. A shallow lake of large surface area is more difficult to destratify than a deep lake of small surface area. Destratification performance is more efficient in the spring than in the fall. Meteorologic conditions such as temperature, sunlight, and wind speed and direction, would affect the calculated DE for any given mixing operation. The elapsed time a destratification device may be operated is controlled by water quality considerations, not the change in temperature profile. Therefore, for any given mixing operation, the mixing device might be run past the point of optimum hydraulic performance because the impoundment's water quality needed further improvement. This, of course, is the proper method of operating a destratification device, but it may influence the calculated DE to give any particular mechanical device a low rating. In the King George VI experiment,[18] horizontal circulation in the rectangular reservoir was another feature of the mechanical pump operation. Additional energy beyond that required for vertical mixing was supplied. The data shown in the last two columns of Table 4 are useful in giving a general concept of the energy needed for mixing a body of water, but any installation may deviate from the figures presented there.

Table 4. COMPARISON OF DESTRATIFICATION EFFICIENCIES OF VARIOUS STUDIES

Lake reference	Dates of mixing	Method	Volume, acre-ft	Change in stability (uncorrected (U) or corrected (C)), kw-hr	Total energy input, kw-hr	Destratification efficiency-DE[a] (calculated over entire mixing period),%	Total energy input/unit volume/mixing,[a] kw-hr/acre-ft
Test Lake 1	6/2-6/7, 6/21-6/24, 7/15-7/20, 8/18-8/23, 1966	Diffused-air pump	2,900	27 (C) 23 (C) 33 (C) 16 (C)	1,760 1,680 2,200 2,860	1.5 1.4 1.5 0.6 1.2[b] 0.5[d]	0.6 0.6 0.7 1.0 0.7[b] –
Test Lake 2	(5/16-5/19, 5/20-5/23), 6/10-6/15, 7/1-7/8, 7/28-8/3, 9/8-9/13, 1966	Diffused-air pump	4,600	7 (C) 24 (C) 35 (C) 28 (C) 8 (C)	3,285 2,740 3,740 3,550 2,540	0.2 0.9 0.9 0.8 0.3 0.6[b] 0.2[d]	0.7 0.6 0.8 0.8 0.5 0.7[b] –
Vesuvius Lake[1]	9/3-9/17/64	Mechanical pump	1,260	6.7 (U)	1,862	0.4[c]	1.5
Boltz Lake[2,3]	8/6-9/10/65	Mechanical pump	2,900	28 (C)	14,300	0.2 0.1[d]	4.9
King George VI Reservoir[18]	7/13-7/20/66	Mechanical pump	18,375	44.3 (U)	2,000	2.2[c]	0.1
Lake Wohlford[17]	4/18-4/25/62	Diffused-air pump	2,500	14.8 (U)	2,570	0.6[c]	1.0
Wahnbach Reservoir[8]	6/9-9/9/64	Diffused-air pump	33,740	400 (U)	104,800	0.4[c]	3.1
Cox Hollow Lake[7]	6/30-7/18/66	Aero-Hydraulics Gun	1,190	7.9 (U)	2,140	0.4[c] 0.6 (avg.)	1.8 2 (avg.)

[a] These data are intended as a rough guide only and are influenced by many factors: shape of lake, time of year, meteorologic conditions (temperature, sunlight, wind) quality requirements, circulation requirements, and so forth.

[b] Average for all mixings in 1966.

[c] These data may be somewhat in error as the change in stability was not corrected for any natural change in stability that may have occurred during the mixing operation.

[d] Calculated by the area method, see text.

STABILITY DECLINE RATE

Another difficulty in calculating the DE for a mixing operation is the nonlinear rate of stability decline with time during mixing. This has been reported upon by Ford[17] and others.[3] For example, during the second and third mixing in Test Lake 2 in 1966, the rate of stability decline decreased as mixing proceeded (Figure 20). During mixing 2, from day 2 to day 5, little change in stability occurred, despite a continued energy input of 449 kilowatt-hours per day. In this case, solar radiation was so strong that the surface was being reheated as quickly as the mixing device distributed this heat throughout the lake. Because the object of any mixing operation is quality improvement, the diffused air system was operated from day 2 to day 5 in an attempt to provide DO in the

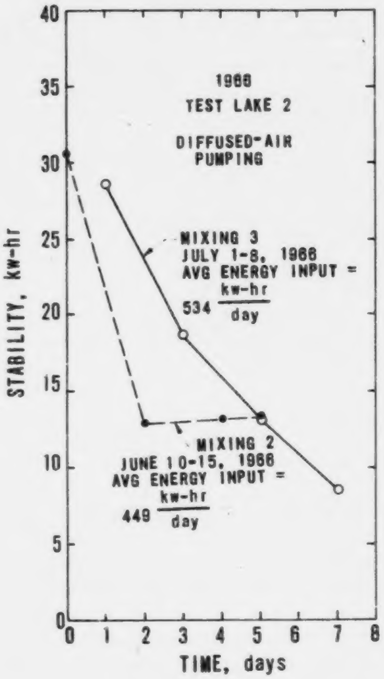

Figure 20. Decline of rate of change of stability with time during a mixing operation.

deeper waters. Table 5 shows the uncorrected DE over short time spans during these two mixings. These data show that any artificial destratification device may produce a low DE indicating poor mechanical performance, when the device was actually being operated beyond the point of maximum hydraulic effectiveness. In both mixing 2 and mixing 3, the overall DE for this device is much lower than the DE calculated for the first portion of the mixing operation.

Table 5. RELATIVE CHANGE IN DESTRATIFICATION EFFICIENCY WITH TIME DURING ONE MIXING OF TEST LAKE 2

Day	Total energy input, kw-hr	Gross decline in stability - uncorrected for natural changes, kw-hr	Uncorrected destratification efficiency, percent
		Mixing 2	
0-2	898	17.9	1.99
2-4	898	-0.2	0
4-5	449	-0.5	0
Overall	2,749	17.2	0.63
		Mixing 3	
1-3	1,067	9.8	0.92
3-5	1,067	5.6	0.52
5-7	1,067	5.0	0.47
Overall	3,200	20.4	0.64

SUMMARY

Comparison of the Two Mixing Systems

The data presented in this paper show that artificial destratification, either by mechanical- or diffused-air pumping, produces a beneficial water quality effect. Any comparison of the two systems, therefore, must be made on a performance and economic basis, rather than a water quality basis.

Because two test variables were changed between the 1965 and 1966 experiments (from mechanical to diffused-air pumping and from a single late summer mixing to multiple mixing beginning in the spring), comparing the two systems is difficult. In general, the diffused-air pump had a better OC and DE than the mechanical pump. The DE comparison may not be fair, however, because of the hydraulic inefficiency of the pump selected for the 1965 test. To quote Thackston:[19]

''The high discharge velocity from the pump (8.4 fps), with its associated high velocity head and hydraulic losses, consume a considerable part of the pump's output. The substitution of a large pump of the same capacity would obviously reduce this velocity head and thus reduce the input power per unit volume actually pumped. However, at the same time, the reduction of discharge velocity would reduce the momentum transferred from the discharge jet to the waters of the epilimnion, and, consequently reduce the mixing, or current-producing potential of the pump.

''Another possible disadvantage of lowering the exit velocity is that mixing of the cold hypolimnion waters with the warmer epilimnion would be inhibited. The resulting temperature of the discharge would tend to remain lower and the water might tend to drop back to the hypolimnion.''

Until a high-capacity, low-velocity mechanical pump is tested, the accuracy of Thackston's comments are unknown. As Thackston notes, considerable energy losses would be avoided by using a lower velocity mechanical pump, but these might well be offset by less efficient mixing of the cold and warm water at discharge and reduced inertial currents. Previous calculations indicated that only 50 percent of the water moved passed through the mechanical pump.[1,2]

The diffused-air system operates as a high-volume, low-velocity pump. If the vertical velocity was 0.1 fps, 250 acre-feet per day would pass through the 40-foot diameter boil cross-section at the bottom of the lake. Very low headlosses would be associated with such a low velocity operation, yet a large quantity of water would be moved. Because the rising water is unconfined, it entrains considerable warmer water during its rise to the surface. Float studies indicated that a strong radial, horizontal velocity away from the boil area occurred approximately 10 feet below the lake surface. This indicates that the rising water has been warmed sufficiently during its movement to the surface, so as to sink only 10 feet, and thus not return to the hypolimnion. This is most advantageous. Whether similar results could be obtained with a mechanical pump can be determined only by testing this type of apparatus.

Judging Hydraulic Performance

Judgment of the hydraulic performance of artificial destratification devices by comparison of their DE may be misleading because the endpoint of most mixing operations should be controlled by water quality considerations, rather than by hydraulic performance considerations. For example, in Wirth's experiment,[7] Cox Hollow Lake was nearly completely destratified during the first 20 days of operation. Since then, the Aero-Hydraulics Gun has been operating continuously with a DE of zero with the purpose of maintaining good water quality. By operating this equipment continuously, oxygen has been maintained, ice prevented from forming, and the usual wintertime fish kill avoided. Similar experiences were recorded by Bernhardt.[9] In his experiment, which lasted from June 11 to November 11, 1964, and produced a DE of 0.4 percent, the lake was nearly destratified in two-thirds of the total operating time. Stopping the experiment at this time and calculating the DE for his diffused-air system would have raised it to 0.6 percent.

Although DE and its companion parameter energy input per unit of impoundment volume are useful values for general sizing of equipment, comparisons should be made on an OC or other water quality parameter basis, because water quality improvement is the objective of destratification. For example, Bernhardt[9] devised a method of aerating the Wahnbach Reservoir hypolimnion without changing the temperature profile. His device had a DE of zero but an OC of 2.1 pounds DO per kilowatt-hour. Except for such special cases, most reaerating devices do destratify the body of water and therefore a fairly calculated DE serves some useful purpose.

Proposed Method of Fair Evaluation

To obtain a fair evaluation of the hydraulic performance of a destratification device, the following testing procedure is recommended. A record of the stability pattern in the body of water to be destratified should be available for some years previous to the test. Second, sufficient temperature profile data should be collected just prior to the experiment to determine the stability trend in the lake at that time of the year. Third, after the destratification experiment begins, stability reduction data should be collected as quickly as possible to determine the maximum hydraulic performance of the destratification equipment. Finally, other destratification devices should be tested at the same time of the year so that the natural pattern of stability would be similar during all tests.

Ease of installation and maintenance of operation will influence the choice of mixing equipment. Where an existing lake or reservoir is to be mixed, a diffused-air system has advantages, as the power unit is on shore where installation and servicing are convenient. In a proposed impoundment, high discharge mechanical pumping equipment could be included in the original dam design, making for greater ease of operation.

Reservoir Water Quality

Both methods, mechanical and diffused-air pumping broke up the thermal stratification patterns in the study lakes successfully and improved water quality. In both cases, DO was added to the water and sulfides, reduced iron and manganese were oxidized. Although some nitrogen and phosphorus were transported to the surface layers during mixing, excessive algal growths did not develop in either case. In both cases, actually, some environmental factor or factors changed during mixing, causing algal populations to decline. Populations of blue-green algae declined more than green-algal populations. In neither case were the declines permanent; the algal numbers increased again after mixing stopped.

FUTURE PLANS

Some questions still remain unanswered. The environmental changes that occur causing alteration in algal numbers and type should be investigated. To date (1967), no studies have been made on the behavior of indicator organisms and pathogens in stratified and artificially destratified impoundments. In municipal water supply and recreation reservoirs data of this type are important if the bacterial and viral qualities of the water are to be predictable. Also, the influence of mixing on fish population is not known.

In the area of hydraulic performance, studies should be made to determine whether energy requirements would be equal for energy supplied in a slow, steady manner as compared to a short, intense pulse. The latter would be more economical if off-peak electric power could be used. Finally, studies should be made on the hydraulic performance of a high-discharge, low-velocity mechanical pump.

CONCLUSIONS

Based on the data taken in these experiments, the authors conclude:

1. Both mechanical- and diffused-air pumping will destratify a body of water artificially.

2. Multiple mixing started in the spring will maintain good quality water in an impoundment, and is superior to a single mixing in midsummer or early fall.

3. Under the conditions of this experiment, the diffused-air pump operated better than the mechanical pump.

4. Because the mechanical pump used in this experiment was hydraulically inefficient, diffused-air pumping cannot be judged superior in all cases, on the basis of data presented here.

5. Some rise in nitrogen and phosphorus concentrations occurred in the surface water during artificial destratification, but the surface populations of phytoplankton and the phytoplankton standing crop declined during mixing.

6. Blue-green algal populations declined more during any one mixing than green algal populations.

7. The reason for the change in algal populations is not known; however, the algal populations increased again when mixing stopped.

8. Comparisons of oxygenation capacity of various pieces of equipment are difficult because it is influenced by the condition of the lake at the time of mixing.

9. Comparisons of destratification efficiency of various pieces of equipment should be made at similar times of the year, in similar lakes, during the early stages of any destratification operation.

10. Data taken in this study and from five other studies in the literature, showed a destratification efficiency ranging from 0.1 to 2.2 percent and averaging 0.6 percent,

and an energy input per unit of impoundment water volume ranging from 0.1 to 4.9 kilowatt-hours per acre-foot and averaging 2 kilowatt-hours per acre-foot (Table 4).

ACKNOWLEDGMENTS

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PAPER NO. 13

INFLUENCE OF ARTIFICIAL DESTRATIFICATION ON PLANKTON POPULATIONS IN IMPOUNDMENTS

E.L. Robinson, W.H. Irwin, and J.M. Symons

This paper, the seventh in the series of eight on artificial destratification for reservoir water quality control, reports in detail on the plankton data taken during the 1965 and 1966 mixing experiments. The total algal counts, the counts of various types of algae—green, blue-green, and so forth—and the number of occurrences of various species of algae are reported.

The correct reference citation is:

Robinson, E. L., W. H. Irwin, and J. M. Symons. "Influence of Artificial Destratification on Plankton Populations in Impoundments." Transactions of the Kentucky Academy of Science, 30, Nos. 1&2, 1-18. 1969

INTRODUCTION

Recently, much attention has been given to the influence of thermal stratification on water quality in reservoirs and the possible improvement of reservoir water quality by artificial destratification. During the past several years, the authors have investigated water quality in three northern Kentucky fishing lakes to determine the effectiveness of controlling water quality by artificial destratification. Previous papers on this subject,[1,2,3,4] have mainly discussed the influence of mixing on chemical water quality.

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The purpose of this paper is to show the phytoplankton populations that existed during destratification. Enumerations of plankters were made of total and generic groups existing before and after mixing. Also, the results of both years' studies were compared to determine any similarities or differences.

LAKES STUDIED, EQUIPMENT, AND ANALYTICAL METHODS USED

The control lake, Bullock Pen Lake, was not mixed, but was used to show the general trend of water quality characteristics for lakes in the area studied. Test Lake 1, Boltz Lake, was artificially destratified by mechanical pumping continuously for 36 days from August 6 to September 10, 1965, and by diffused-air pumping for four periods totaling 14.4 days between June 3 and August 23, 1966. Test Lake 2, Falmouth Lake, studied only in 1966, was destratified by the diffused-air system for five periods totaling 28.5 days between May 16 and September 13, 1966. The morphological characteristics of the study lakes during the 1965 and 1966 experiments, and the mechanical pump used for destratification have been described previously.[1,2] The diffused-air system has also been described previously.[3,4] Both mixing devices were placed at or near the deepest portions of the lakes studied.

Water samples were taken at various depths and times in the deep-water areas to determine the influence of destratification on water quality. Details of sampling techniques and methods of field and laboratory analyses for physical and chemical constituents of the study lakes have been described previously.[4] One-liter water samples were collected from Bullock Pen Lake to study the influence of artificial destratification on plankton populations at the surface and at depths of 5, 15, 30, and 43 feet; at the surface and at depths of 5, 15, 30, 45, and 60 feet from Boltz Lake; and at the surface and at depths of 5, 15, 30, and 40 feet from Falmouth Lake. Samples were preserved with 0.16 percent thimerosal and 1 percent Lugol's solution. One gram of sodium borate was added for each gram of thimerosal to keep the thimerosal in solution. This preservative, reported by Williams,[5] aided preservation and precipitation of organisms for microscopic identification.

Samples were examined in a Sedgwick-Rafter cell with a 20-power objective and 10-power oculars. The ''strip counting'' method was used; calibration, dilution factors, and the number of plankters per milliliter were computed.

All recognizable plankters were identified to genus and enumerated. Each single-cell organism was counted as one, and organisms grouped in colonies were counted as one. Filamentous colonies were listed as units of 100 microns in length, and only whole zooplankters were enumerated. No special effort was made to identify the plankters to species. Diatoms were identified only from wet mounts, and single cells of less than 4 microns were not enumerated. Some samples contained so few plankters that concentration was necessary. Aliquots of 200 and 500 milliliters, depending upon the concentration of plankters, were filtered through membrane filter discs (0.4 micron). Discs were placed in 250-milliliter beakers and gently rinsed with distilled water. The rinse water was used to make a slurry that was diluted to a desired volume. The discs were examined under 430 X magnification to establish that the organisms had been removed by this technique. The concentrated organisms were then examined in a Sedgewick-Rafter cell.

RESULTS

Total Plankton Population

Plankton populations occurring in the three study lakes during 1965 and 1966, and the average plankton count throughout the total depth of each lake were computed for comparison. The plankton numbers per milliliter at the 5-foot depth were assumed to represent the organisms in the 0- to 5-foot layer; the 15-foot count was assumed to represent the organisms in the 5- to 15-foot water layer, and so forth. The plankton counts per milliliter at the various depths were multiplied by the water volume of the layers they represented, and the arithmetic sum for the entire lake was divided by the total volume of the lake to obtain the average number of plankters per acre-foot throughout the lake.

The total plankton counts at the 5-foot depth in the unmixed lake for 1965 and 1966 were compared (see Figure 1) to show similarity. The data in Figure 2 show the similarity of plankton in the control lake and Test Lake 1 in 1965 during the months before mixing began. A sharp decline in the plankton count occurred when the artificial destratification began, and the decline continued while pumping continued in Test Lake 1. The plankton count also

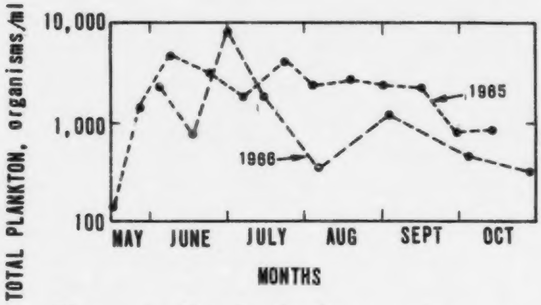

Figure 1. Total plankton counts at the 5-foot depth in the unmixed lake.

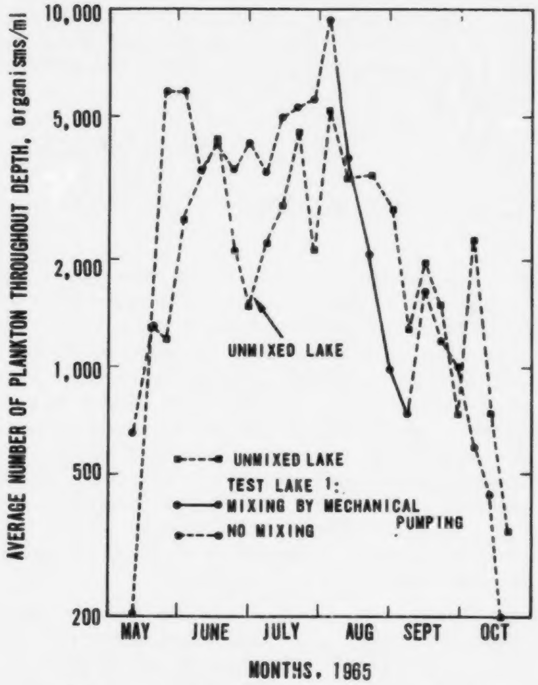

Figure 2. Behavior of total plankton population in unmixed lake and Test Lake 1 during 1965.

declined in the unmixed lake at this time, but the decline was less rapid than that in Test Lake 1.

Figure 3 shows sharp decreases in the total plankton counts during mixings 2, 3, and 4, in Test Lake 1 in 1966, although the decreases were only temporary. A somewhat different growth pattern occurred in Test Lake 2 during 1966, as shown in Figure 4. The total plankton numbers increased during mixings 1, 2, and 5. Small declines were recorded during mixings 3 and 4.

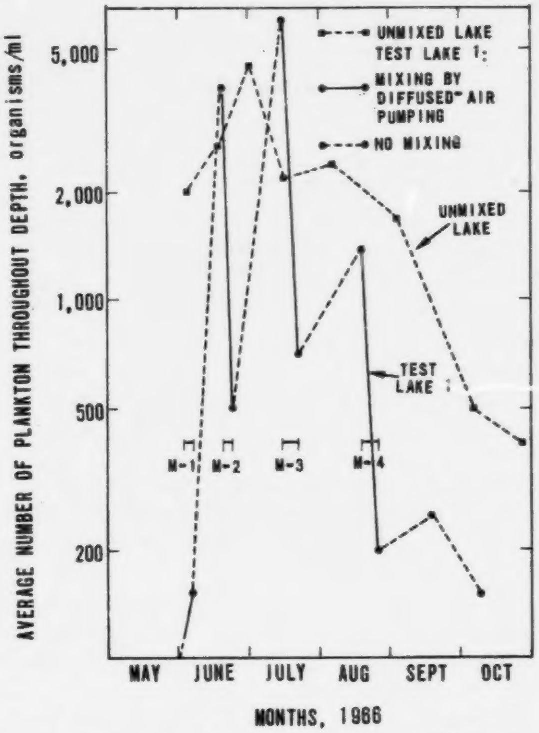

Figure 3. Behavior of total plankton population in unmixed lake and Test Lake 1 during 1966.

Plankton Distribution With Depth

The plankton concentration varied with depth in all three lakes. The unmixed lake, as an example, is shown

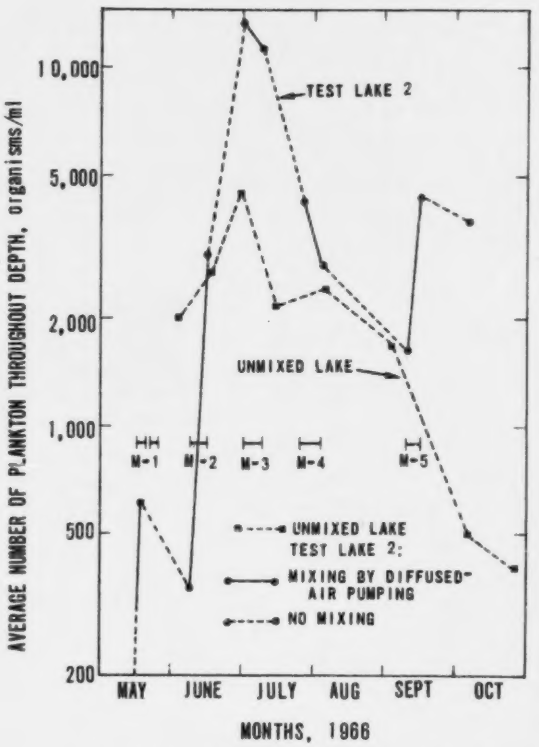

Figure 4. Behavior of total plankton population in unmixed lake and Test Lake 2 during 1966.

in Table 1. As the summer season progressed the plankton concentration was highest in the lower layers. This lake was sufficiently clear to support plankton populations throughout its total depth.

Table 2 demonstrates that, in 1966, in Test Lake 1, during mixings 2, 3, and 4, the plankton numbers declined at all levels. This shows that the surface algae were not transported to the lower layers in the lake during mixing. Test Lake 2 behaved slightly differently from Test Lake 1, probably because of an increased phosphorus concentration near the surface. In this case, although the data from all depths showed similar trends, the plankton counts in the 5-foot samples were consistently and significantly higher than those taken at the other depths.

Table 1. TOTAL PLANKTON CONCENTRATIONS AT VARIOUS DEPTHS IN UNMIXED LAKE IN 1966

Date	Total plankton count/acre-foot (10^9 omitted after each value) 0 to 5-ft layer	5 to 15-ft layer	15 to 30-ft layer	30 to 44-ft layer
June 3	3.1	1.4	1.1	0.9
June 16	1.5	1.1	5.3	7.3
June 29	4.1	9.5	3.4	4.3
July 13	2.4	2.5	2.3	3.7
Aug. 5	0.4	0.9	7.2	5.8
Sept. 3	1.3	1.3	1.8	3.6
Oct. 5	0.5	0.2	1.1	1.0
Oct. 27	0.4	0.4	1.2	0.7

Table 2. TOTAL PLANKTON CONCENTRATIONS AT VARIOUS DEPTHS IN TEST LAKE 1 IN 1966

Date	Total plankton counts/acre-foot (10^{10} omitted after each value) 0 to 5-ft layer	5 to 15-ft layer	15 to 30-ft layer	30 to 45-ft layer	45 to 62-ft layer
May 23	1.4	0.6	0.7	1.7	0.7
June 7 (A-1)[a]	2.0	2.4	1.1	3.6	0.8
June 21 (B-2)	87.0	45.0	6.4	14.0	15.0
June 25 (A-2)	9.1	5.5	3.7	3.7	1.3
July 13 (B-3)	140.0	51.0	51.0	27.0	36.0
July 20 (A-3)	5.9	7.7	9.6	6.1	14.0
Aug. 17 (B-4)	19.0	21.0	33.0	9.0	3.7
Aug. 23 (A-4)	0.9	2.0	1.3	1.2	2.4
Sept. 16	3.4	3.0	2.8	2.9	3.1
Oct. 5	0.8	1.9	1.7	2.5	1.5

[a] A-1 means after mixing 1; B-2 means before mixing 2; and so forth.

Occurrence of Plankton Categories in Surface Populations

The green and blue-green algae counts at the 5-foot depth in the unmixed lake were similar during 1965 and 1966, as shown in Figure 5. In Test Lake 1 in 1966 the decline in the blue-green algae counts was greater than that in the green algae counts following mixings 2, 3, and 4 (Figure 6). The blue-green algae decrease was more pronounced in 1966 than in 1965.

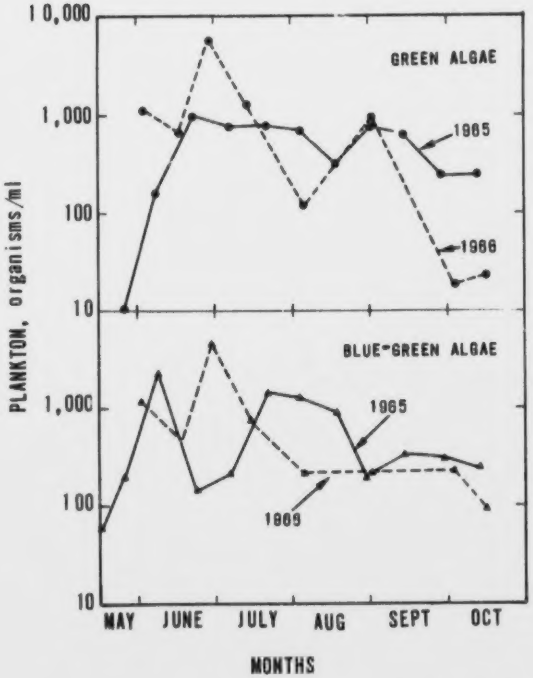

Figure 5. Green and blue-green algae counts at the 5-foot depth in the unmixed lake.

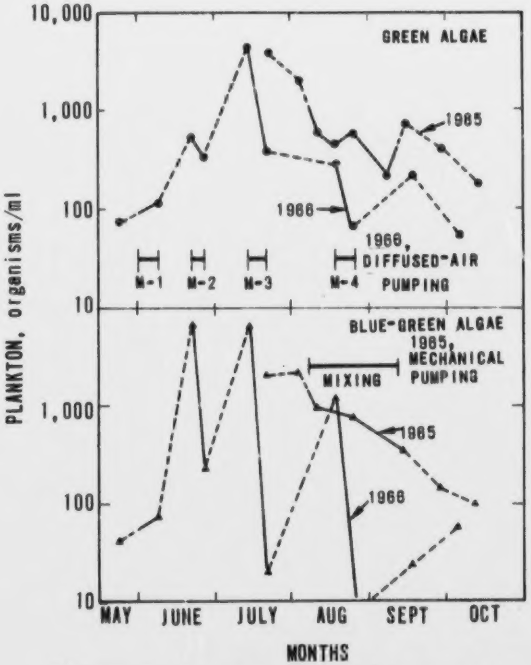

Figure 6. Green and blue-green algae counts at the 5-foot depth in Test Lake 1, 1965 and 1966 (-----no mixing, ——— mixing).

In Test Lake 2 a different pattern was evidenced, as shown in Figure 7. During mixings 1 and 2, the counts of both green and blue-green algae increased, the green algae increase being the greater. During mixings 3 and 4, a slight increase in green algae numbers occurred, while the blue-greens declined. Finally, both categories of algae decreased during mixing 5. Figures 6 and 7 indicate that artificial destratification is more detrimental to blue-green than to green algae.

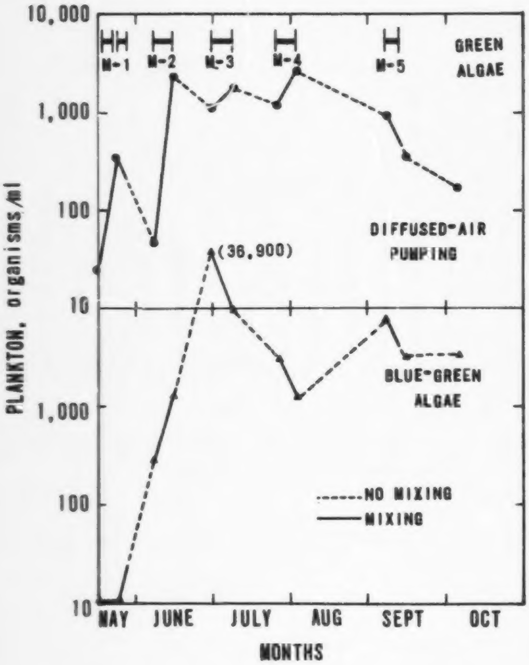

Figure 7. Green and blue-green algae counts at the 5-foot depth in Test Lake 2, 1966.

Composition of Standing Crop

The counts from all samples were separated into five categories: green algae, blue-green algae, diatoms, zooplankton, and pigmented flagellates. The percentage of the total plankton numbers throughout the lake was then calculated for each category. For representation, the zooplankton and the pigmented flagellates were summed to form one category. Figures 8 and 9 show that, in the unmixed lake, the blue-green algae predominated during the months of July, August, and September. Figure 10 shows that, when mixing began in Test Lake 1 in early August 1965, the general upward trend of the blue-green algae stopped and reversed, and green algae predominated during the remainder of that season. The percentage of zooplankton and pigmented flagellates rose sharply during the mixing period.

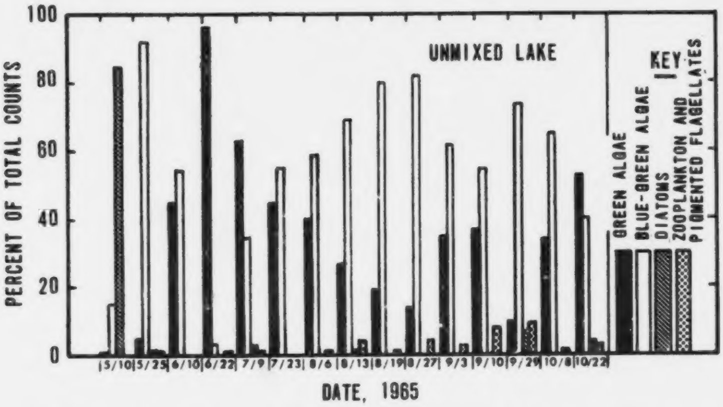

Figure 8. Composition of plankton standing crop in unmixed lake 1965.

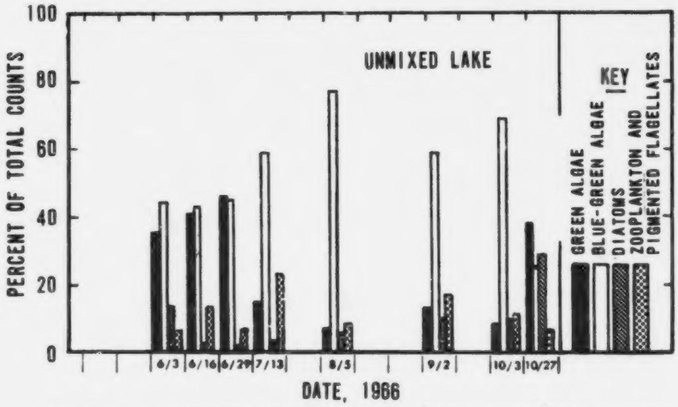

Figure 9. Composition of plankton standing crop in unmixed lake 1966.

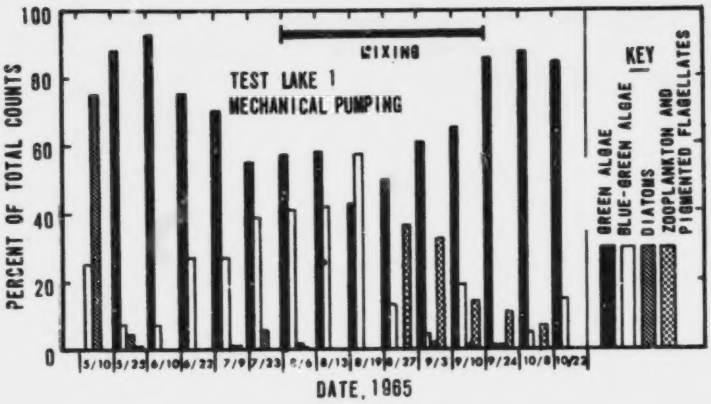

Figure 10. Composition of plankton standing crop in Test Lake 1 during 1965.

Figure 11 shows that in Test Lake 1 in 1966, mixings 2, 3, and 4 resulted in a sharp decrease in the percentage of blue-green algae. Diatoms increased during mixing 2, and the combined zooplankton and pigmented flagellates increased during mixing 3.

A different situation developed in Test Lake 2 in 1966, as shown in Figure 12. During mixings 2, 3, and 4, there was a reduction in the percentage of blue-green algae,

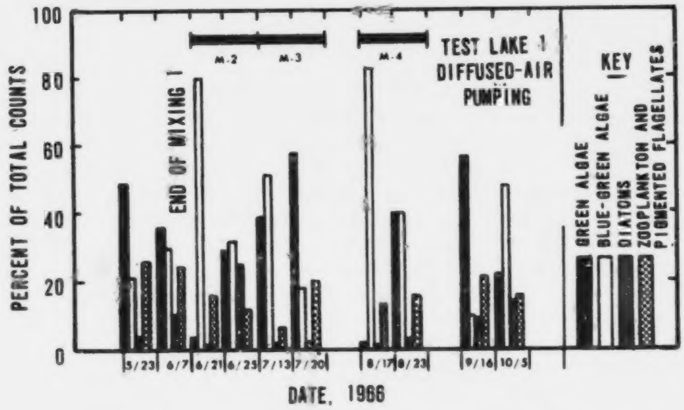

Figure 11. Composition of plankton standing crop in Test Lake 1 in 1966.

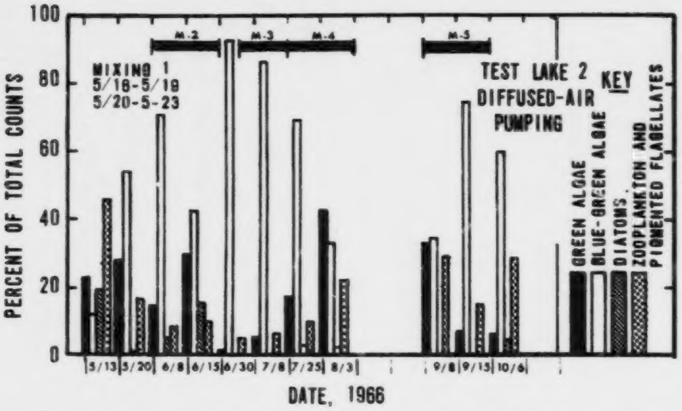

Figure 12. Composition of plankton standing crop in Test Lake 2 in 1966.

although it was not as pronounced as in Test Lake 1 in 1966. During mixing 5, the percentage of blue-green algae increased, probably because blue-green algae tend strongly to predominate in September in this geographical location. Also, Figure 12 shows a high percentage of the combined zooplankton and pigmented flagellates category. The environment in Test Lake 2 differed somewhat from Test Lake 1, supporting a wider variety of organisms. Changes in plankton composition during artificial destratification have also been noted by Bernhardt[6] and by Cooley et al.[7]

Number of Plankton Genera

The totals of the different plankton genera in the standing crop are presented in Table 3. There was an increase in the unmixed lake as the water temperature increased. Test Lake 1 had little or no change in the number of genera during mixings 2, 3, and 4. Test Lake 2 showed significant increases in the number of plankton genera during mixings 2 and 3, but only slight increases during mixings 4 and 5. Possibly the warming of the water during early summer in Test Lake 2 permitted plankton to grow in wider variety.

Predominant Organisms

During the 2 years of study, 81 genera of algae were identified from the three lakes. From the unmixed lake, 56 genera were identified, 33 of them found in both years.

Table 3. NUMBER OF PLANKTON GENERA IN THE STANDING CROP ON VARIOUS DATES IN 1966 IN THE THREE STUDY LAKES

Unmixed lake		Test Lake 1		Test Lake 2	
Date	Number of genera	Date	Number of genera	Date	Number of genera
				5/13	25
		5/23	18	5/20	27
6/3	18	6/7	27	6/8	16
6/16	21	6/21	25	6/15-2[a]	27
6/29	29	6/25-2	25	6/30	17
7/13	33	7/13	36	7/8-3	24
		7/20-3	41	7/25	23
8/5	27			8/3-4	28
		8/17	30		
		8/23-4	30		
9/2	33			9/8	32
		9/16	38	9/15-5	35
10/3	36	10/5	31	10/6	30
10/27	33				

[a] -2 indicates mixing 2 ended. Mixing 2 started on the previous date. The same code is used for the other mixings.

In Test Lake 1, 63 genera were found, 30 of them in both years. In Test Lake 2, 48 genera were found. Thirty-five genera were common to all three lakes. All plankton genera identified during this study are presented in Table 4. The total number of samples from each study lake and the number of times that a given organism occurred in any sample are presented to show which of the algal genera appeared most frequently. Some data on the occurrence of zooplankton genera are also given in this table.

The predominant plankton, based on the numbers of organisms in the unmixed lake were as follows: green algae - *Ankistrodesmus*, *Chlamydomonas*, and *Chlorella;* blue-green algae - *Anabaena*, *Anacystis*, *Aphanizomenon*, and *Oscillatoria;* diatoms - *Asterionella*, *Nitzschia*, *Melosira*, and *Synedra;* pigmented flagellates - *Ceratium*, *Chrysococcus*, *Glenodinium*, and *Trachelmonas*.

The predominant genera of algae at the time of optimum growth in Test Lake 1 during 1966 were: green algae - *Chlorella* and *Chlamydomonas* (their numbers were higher in the upper layers); blue-green algae - *Anabaena* and *Anacystis*. Pigmented flagellates appeared for a short period, the most abundant being *Chrysococcus*, *Euglena*, and *Ceratium*. Few zooplanktonic forms were found in Test Lake 1 during 1966.

Table 4. NUMBER OF TIMES PLANKTON ORGANISMS APPEARED IN SAMPLES

GENERA **Green algae Non-filamentous**	Bullock Pen 1965 (66 samples)	Bullock Pen 1966 (32 samples)	Beltz 1965 (120 samples)	Beltz 1966 (50 samples)	Falmouth 1966 (44 samples)
Ankistrodesmus	2	5	–	8	8
Botryococcus	–	–	1	–	–
Chlorobotrys	1	–	–	–	–
Chlorella	5	8	5	10	9
Chlorogonium	–	–	1	–	–
Chlorococcum	–	4	–	10	4
Chodatella	–	1	–	–	4
Chroococcus	–	1	–	1	–
Closterium	3	4	9	6	10
Coelastrum	19	8	19	9	7
Cosmarium	3	7	2	5	1
Crucigenia	–	–	–	2	1
Dictyosphaerium	2	1	11	7	1
Eremosphaera	–	1	–	–	–
Eurastrum	–	2	–	2	–
Franceia	–	1	–	–	3
Gleocystis	1	2	1	8	4
Golinkinia	–	1	5	–	1
Hormidium	–	–	1	–	–
Lagerheimia	–	–	–	–	1
Oöcystis	17	8	19	10	10
Pamella	–	2	4	4	–
Pandorina	–	–	–	–	2
Pediastrum	3	1	2	1	–
Planktosphaeria	–	1	–	–	–
Pleodorina	1	–	–	–	–
Seenedesmus	7	7	5	7	8
Selenastrum	–	–	–	–	1
Sphaerocystis	17	2	8	–	2
Staurastrum	8	2	12	7	2
Sticococcus	–	–	–	2	–
Tetraedron	–	3	–	–	–
Green algae filamentous					
Cladophora	–	1	–	1	–
Hormidium	1	–	1	–	–
Microspora	1	–	–	1	–
Oedogonium	–	–	–	–	1
Spyrogyra	–	–	1	1	–
Stigeoclonium	–	1	–	–	–
Tribonema	1	1	–	2	–
Ulothrix	–	1	–	–	–
Green algae flagellates					
Chlamydomonas	23	8	21	10	11
Coccomonas	–	3	–	2	2
Gonium	–	1	–	2	4
Phacotus	9	2	10	8	9

GENERA	Bullock Pen 1965 (66 samples)	Bullock Pen 1966 (32 samples)	Beltz 1965 (50 samples)	Beltz 1966 (50 samples)	Falmouth 1966 (44 samples)
Blue-green algae					
Anabaena (100µ)	12	8	17	10	11
single cells	9	–	10	–	–
Anacystis	14	8	–	10	9
Aphanizomenon	14	4	12	7	5
Coelosphaerium	–	8	–	2	–
Cylindrospermum	1	–	2	7	10
Gomphosphaeria	9	4	–	–	–
Oscillatoria	13	8	11	7	8
Phormidium	–	2	–	1	5
Pigmented flagellates					
Ceratium	8	6	10	7	6
Chrysococcus	4	5	3	5	11
Cryptomonas	4	4	–	6	10
Dinobryon	–	–	–	1	–
Euglena	5	8	4	6	11
Euglenopsis	1	–	–	–	–
Glenodinium	–	–	–	5	7
Golenkinia	–	–	6	–	–
Gymosinium	1	–	–	–	3
Lepocinclis	–	–	–	–	1
Mallomonas	4	3	6	2	4
Peridinium	5	6	1	1	5
Phacus	–	1	1	5	6
Trachelmonas	17	8	11	7	11
Diatoms					
Achnanthes	–	1	–	4	7
Asterionella	5	7	8	9	1
Cyclotella	1	3	1	2	4
Cymbella	–	2	–	2	2
Fragillaria	2	3	–	6	6
Gomphonema	–	–	–	2	3
Melosira	–	6	–	3	5
Navicula	1	2	–	3	2
Nitzschia	2	5	1	1	4
Pennularia	–	1	–	–	1
Rhopalodia	2	1	–	–	–
Stauroneis	2	3	1	1	2
Synedra	2	5	1	5	5
Zooplankton					
Crustaceans					
Copepods	4	3	4	5	2
Cladocerans	3	1	2	7	2
Rotifers					
sp.	5	–	–	–	2
Keratella	5	2	4	7	3
Ciliophorans sp.	6	–	6	4	4
Rhyzopods sp.	1	2	1	5	–
Actinopods					
sp.	1	–	–	–	–
Codenella	–	–	–	–	5
Nematodes sp.	–	1	–	–	1

In Test Lake 2 in 1966, the greatest number of genera was among the green algae but the largest populations were blue-greens. At the midsummer optimum, the most abundant genera of the blue-green algae were *Anabaena*, *Anacystis*, *Cylindrospernum*, and *Oscillatoria*. *Phormidium* succeeded in the period following the midsummer optimum and was dominant the remainder of the season. Pigmented flagellates increased as temperatures rose, the predominant genera being *Chlamydomonas*. Also, appreciable numbers of *Glenodinium* and *Trachelmonas* were present.

DISCUSSION

Environmental factors important to plankton production are variable, and natural growth patterns must be compared with those in controlled laboratory experiments to explain the controlling factors and their functions in the algal environment.

Influence of Stratification and Light

The numbers of plankton in the stratified control lake were greatest in the deeper layers. Water of this lake was clear, and its light penetration greater than in the test lakes, effecting increased algal growth. Light intensity was not as great at the lower depths as at the surface; also the plankton may have been stimulated by chemicals concentrated in the stratified layers.

These findings concur with those of Juday and Schomer[8] who demonstrated that excessive light inhibits algal growth. The effect of excessive light intensity on plankton growth is a photo-oxidative destruction of chloroplast enzymes.[9,10] Light intensity also affects the uptake of phosphorus and influences the ratio of phosphorus to sulfur within the algal cells during metabolism.[11] Variances in light intensity may also bring about different requirements for iron and manganese by algae.[12] The complex interrelationship of temperature and light affects carbon dioxide assimilation, thus affecting growth.[13]

Temperature and Turbulence

In all three lakes, blue-green algae predominated when the temperature was the highest. The predominant genera found were similar to those reported by Cole[14] for a southern Kentucky stratified artificial impoundment. The effect of temperature on genera was clearly observed from data of the unstirred lake, which showed blue-greens to be predominant in upper waters where the temperature was above 15°C, and greens predominant in the lower waters where temperatures were less than 15°C. Green algae, however, never were found in quantities sufficient to cause conspicuous blooms.

The effect of increased temperature on green and blue-green algae was demonstrated in laboratory experiments[15] in which a species of *Chlorella* was killed exponentially when heated for short periods, but the blue-green *Anacystis* was able to withstand longer periods of heating. While the waters studied in nature during this experiment did not reach temperatures as high as those in the aforementioned experiment, some correlation may exist.

Chu[16] reported that temperature within normal limits had little effect on the planktonic forms he studied; he stated further that chemical composition of growth medium under suitable conditions of illumination is probably of greatest importance in plankton growth. Powell et al.[17] conducted experiments to assay the effect of turbulent flow on the growth of algae in a closed system. The experiments were arranged to provide the alternating periods of light and darkness reportedly necessary for optimum growth. The experimenters concluded that turbulence alone can be expected to produce favorable light and dark cycles for only a small percentage of algae in a flowing solution. This may account for the failure of blue-green algae to produce blooms in the mixed waters.

Influence of Mixing

Mixing alters the length of time organisms remain in contact with salts, gases, and other chemicals in solution. In spite of the mixing of phosphorus throughout the test lakes, the number of plankters decreased throughout the whole lake in all layers. Many other environmental changes undoubtedly influenced the changes in plankton populations that occurred during mixing. Single species may have been affected in many ways, even increased, but overall the number of algae decreased.

Mixing of waters may move plankters from phototrophic levels to areas of reduced light, altering the time of plankter exposure to illumination and thereby retarding growth.[13]

Nitrogen, Phosphorus, and Trace Nutrients

The nitrogen contents in the three study lakes were consistent with amounts found in nature. Some increase in ammonia-, nitrite-, nitrate-, and organic-nitrogen concentrations occurred in Test Lake 1 during 1966 following each stirring, and some decrease occurred in all but the organic nitrogen concentration in Test Lake 2 during the same summer.

The organic nitrogen concentration may have increased because of extra-cellular substances released by some algae as they lysed or were broken apart by stirring. Growth conditions could have been such that toxins were formed in some species and released into the surrounding waters, thus limiting other species.[18,19,20] Algae are inhibited when the concentration of nitrogen and phosphorus exceeds 20 milligrams per liter,[16] a concentration that was never reached in the three lakes studied. Deficiencies in growth occur when the nitrogen concentration is below 0.2 milligram per liter and when phosphorus is below 0.05 milligram per liter.[21] These levels, also, did not occur in the study lakes; therefore, these two chemicals probably did not limit the whole plankton crop.

Large populations of diatoms require higher concentrations of nitrogen and phosphorus than are generally found in nature, which may explain the scarcity of diatoms in the three lakes studied. Also, some diatoms seem to require a growth factor, which affects the phosphorus dependency.[21] The diatom population increased as the phosphorus concentration increased in Test Lake 2 in 1966. Iron, manganese, and sulfides were found in the study lakes in quantities that probably were not limiting.[22]

Gerloff and Skoog[12] concluded from laboratory studies that available nitrogen is the most important limiting factor in algal growth. This may not always be the case because algal growth in natural waters must depend upon depth, location, time of year, rates of absorption, rates of water replacement, temperature, and other conditions.

CONCLUSIONS

Based on the data presented herein, the following conclusions are drawn:

1. The total plankton numbers declined during most mixings, both in 1965 and 1966.

2. Except for a somewhat higher surface concentration in Test Lake 2 (Falmouth Lake), the plankton densities were similar at all depths in all three study lakes.

3. During most mixings, the blue-green algae populations declined more than the green algae populations.

4. During artificial destratification, the number of plankton species remained the same or increased slightly in both test lakes.

ACKNOWLEDGMENTS

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PAPER NO. 14
CONTROL OF RESERVOIR WATER QUALITY BY ENGINEERING METHODS

J.M. Symons, W.H. Irwin, and G.G. Robeck

This paper, the last in the series of eight on artificial destratification for reservoir water quality control, reports in detail the influence of mixing on 18 water quality parameters. The parameters measured were temperature, concentrations of dissolved oxygen, sulfide, iron, manganese, chemical oxygen demand, ammonia nitrogen, nitrate nitrogen, organic nitrogen, dissolved phosphorus, phytoplankton, hydrogen ions (pH), carbon dioxide, calcium, alkalinity, magnesium, and sulfate, and conductivity. When this paper was presented at the Specialty Research Conference, the emphasis was on methods for designing artificial destratification equipment rather than on the influence of artificial destratification on water quality. The Appendix of this paper contains; 1) a summary of the oral presentation on the subject of design, 2) the formal discussion prepared by Professor Edward L. Thackston, and 3) the oral questions from the audience and responses that followed both formal presentations.

The correct reference citation is:

Symons, J. M., W. H. Irwin, and G. G. Robeck. "Control of Reservoir Water Quality by Engineering Methods." In: Proceedings of the Specialty Conference on Current Research Into the Effects of Reservoirs on Water Quality. R. A. Elder, P. A. Krenkel, and E. L. Thackston, Eds. Tech. Rep. No. 17. Dept. Environmental and Water Resources Engineering, Vanderbilt Univ., Nashville, Tenn. 1968. pp. 335-90.

INTRODUCTION

Controlling the quality of water within a reservoir, rather than attempting to treat and improve the quality of the water as it is withdrawn from the reservoir, prevents downstream water quality problems and assures the usefulness of the entire water volume in the reservoir at all seasons of the year. Research on the development of economical and effective engineering methods for the control of reservoir water quality has been conducted by the authors for the past 3 years. To date, 1967, six small lakes of varying volume, which had a history of water quality deterioration because of thermal stratification, have been artificially destratified by various techniques, at various times of the year, and with varying mixing schedules. The results of these studies, which show a beneficial effect on water quality from destratification without any accompanying detrimental effect, are detailed in References 1 through 7.

The purpose of this paper is to show the influence of artificial destratification as an engineering method of water quality control on all of the 18 water quality parameters that were measured during the most recent, and most successful, artificial destratification demonstration. These parameters in order of their presentation are temperature, concentrations of dissolved oxygen (DO), sulfide, iron, manganese, chemical oxygen demand (COD), ammonia nitrogen, nitrate nitrogen, organic nitrogen, dissolved phosphorus, phytoplankton, hydrogen ions (pH), carbon dioxide, calcium, alkalinity, magnesium, and sulfates, and conductivity.

LAKES STUDIED, EQUIPMENT, AND ANALYTICAL METHODS USED

The morphological characteristics of the lakes studied during the 1965 and 1966 experiments, and the 1966 mixing dates are shown in Table 1; the lake outlines are shown in Figure 1. Bullock Pen Lake, which was not mixed, served to indicate the general trend of water quality characteristics of lakes in the northern Kentucky area where the destratification experiments took place. Boltz Lake was artificially destratified by diffused-air pumping four times in 1966.

The diffused-air pump, previously described in detail,[6] was a 22.4-kilowatt portable, electrically driven, rotary

Table 1. MORPHOLOGICAL CHARACTERISTICS OF STUDY LAKES AND DATES OF TESTING

Name	Volume, acre-ft	Maximum depth, ft	Average depth, ft	Surface area, acres	Dates of mixing, 1966
Bullock Pen Lake	3,200	48	23	142	
Boltz Lake	2,900	62	30	96	June 2-7 June 21-24 July 15-20 Aug. 18-23

air compressor attached to a system of diffuser stones by air tubes designed to carry the compressed air to the lake bottom for release. The diffuser system was placed about 300 feet from the dam in Boltz Lake and at the greatest depth, 62 feet.

Water quality samples were taken at weekly intervals at various depths near the dam in Bullock Pen Lake and near the mixing site in Boltz Lake from May to August in

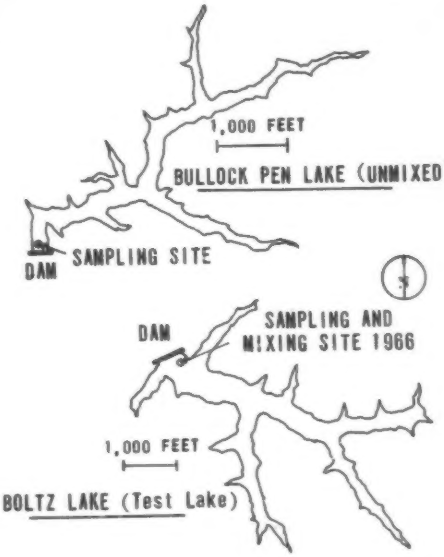
1,000 FEET
BULLOCK PEN LAKE (UNMIXED)
SAMPLING SITE
DAM
DAM
SAMPLING AND MIXING SITE 1966
1,000 FEET
BOLTZ LAKE (Test Lake)

Figure 1. Outlines of unmixed and test lakes.

1965. The purpose of this sampling was to show that these two lakes, which are approximately 10 miles apart, behaved similarly when not mixed. In 1966, water quality samples were taken from Boltz Lake near the mixing site before and after each mixing, and at various depths, to determine the changes caused by mixing. For purposes of comparison, samples were taken on the same, or nearly the same, dates in the unmixed lake, Bullock Pen.

Of the 18 analyses performed on each sample, 6 were performed in the field, and the remaining 12 were performed on samples returned to the laboratory. The details of several of the analytic techniques have been described previously, and are noted herein by reference only. Analytic techniques used in the field are described as follows: for DO, the Winkler Method[1]; for temperature, by thermistor[2]; and for carbon dioxide and alkalinity, by titration.[5] pH was determined with a Beckman Model-G, and conductivity was determined with an Industrial Instrument Inc., Model RB-3.

The laboratory analytic techniques for sulfide, manganese, and dissolved acid hydrolyzable phosphorus,[*] from *Standard Methods*[8] are described in Reference 2; those for ammonia nitrogen, nitrate nitrogen, and total organic nitrogen by automatic analysis[9]; and for phytoplankton density by the Sedgwick-Rafter method.[6] Total iron was determined by a Technicon AutoAnalyzer, using 2,4,6-Tripyridyl-s-Triazine[10] as the indicator.

The concentration of chemical oxygen demand (COD) was determined following the method outlined in *Standard Methods*.[8] The concentrations of calcium and magnesium were determined by finding the total hardness of a sample using the EDTA Titrimetric method,[8] and then determining the calcium concentration using the EDTA Titrimetric method with muroxide as the indicator.[8] The magnesium concentration was then determined by difference. The concentration of sulfate was determined by the turbidimetric method as presented in *Standard Methods*.[8]

RESULTS

An exact control is unobtainable in a field experiment such as this, but, in an attempt to contrast the behavior of water quality in the artificially destratified lake with the general behavior of water quality in an undisturbed lake in the same geographic area, two lakes near each other were chosen for study. Upper water quality was compared by sampling each lake at the 5-foot depth. Lower water quality was reflected by the samples taken at the depth that was 0.7 of the maximum depth: 30 feet in Bullock Pen Lake and 45 feet in Boltz Lake. Although samples were taken at the ''bottom'' of each lake, sampling techniques could influence the nature of the sample actually taken. Therefore, because of the possible unreliable nature of these samples, they were not chosen for comparison.

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[*] Dissolved acid hydrolyzable phosphorus has an operational definition. After centrifugation the supernatant liquor was autoclaved for 20 minutes at 20 psi in the presence of the acid reported in *Standard Methods*.[8] All of the orthophosphate and polyphosphate and some of the organic phosphate was measured. This was arithmetically converted to equivalent phosphorus.

Similarity of Study Lakes When Not Mixed

Although the temperature at the 0.7 depth in Boltz Lake was slightly lower than that in Bullock Pen, the data presented in Figures 2 and 3 indicate that the behavior of water quality in Bullock Pen is also indicative of the water quality in Boltz Lake. Therefore, in the remaining figures, the Bullock Pen data are presented as a ''control'' on the experiment and will be contrasted with the data taken in the lake that was artificially destratified.

The results from the 1966 artificial destratification experiment are divided into five groups of related parameters: (1) temperature and stability; (2) DO, sulfide, iron, manganese, COD, and oxygen balance; (3) ammonia, nitrate, organic nitrogen, dissolved acid hydrolyzable phosphorus, and phytoplankton density; (4) pH, carbon dioxide, alkalinity, calcium, and magnesium; and (5) sulfates, conductivity, and particulates.

Temperature and Stability

Figure 4 shows that the upper water temperature declined with each mixing, but that there was no permanent lowering of upper water temperature in the mixed lake. The temperature at the 0.7 depth, however, showed a rise with each mixing and a general and permanent warming when contrasted to the lower water temperatures of the unmixed lake.

Stability, as a measure of the effectiveness of destratification equipment, has been fully discussed.[2,3,4,6] Calculation of stability changes during artificial destratification is a useful evaluation technique. Figure 5 shows that the stability of the mixed lake was sharply lowered with each mixing, and, although there was a

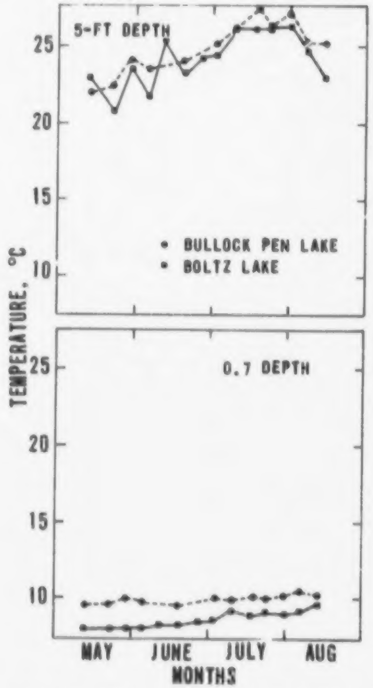

Figure 2. Comparison of Bullock Pen and Boltz Lakes in 1965 - temperature.

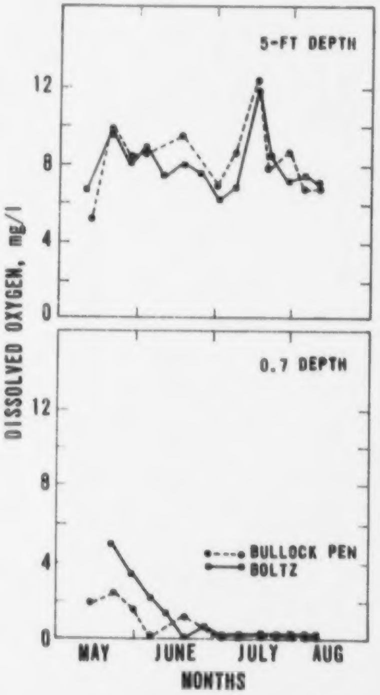

Figure 3. Comparison of Bullock Pen and Boltz Lakes in 1965 -

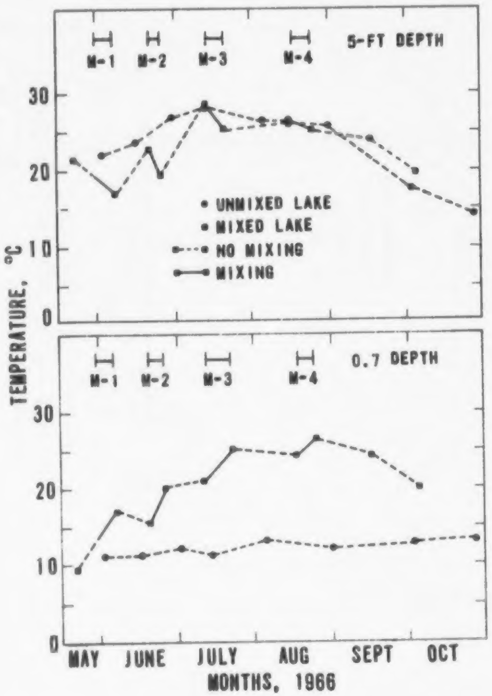

Figure 4. Influence of diffused-air pumping on temperature.

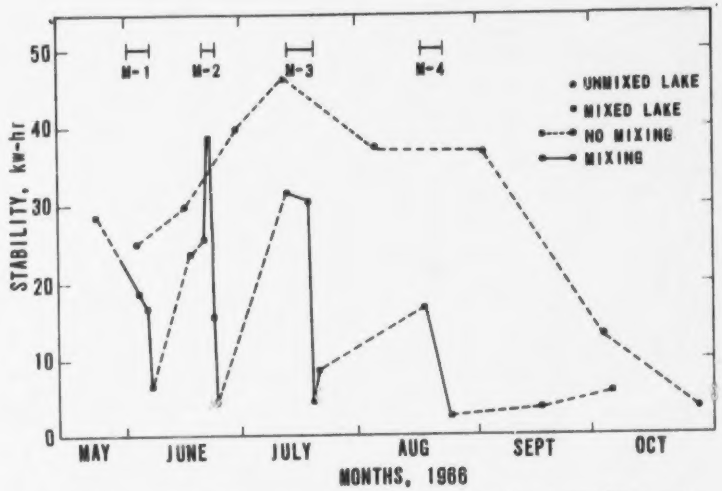

Figure 5. Influence of diffused-air pumping on stability.

tendency for the stability to rise between mixings, the general pattern was distinctly different from the unmixed lake. The rate of rise between mixings declined as the season progressed, however, so mixing could be less frequent in late summer.

DO and Related Parameters

The upper portion of Figure 6 shows a decline in DO concentration at the 5-foot depth, caused by bringing oxygen-demanding materials into the upper layers of the lake during the mixing operation. Photosynthetic algal action and atmospheric reaeration, however, restored the DO to near saturation levels between mixing periods. The lower portion of Figure 6 shows the increase in DO concentrations caused by artificial destratification. The reaeration capacity of the diffused-air pump was not

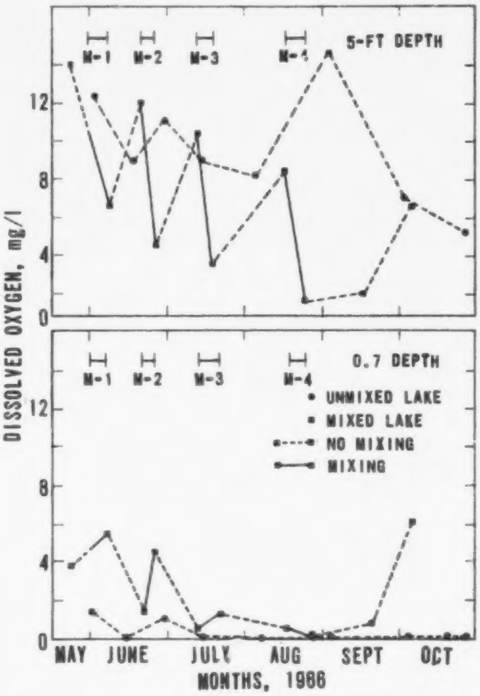

Figure 6. Influence of diffused-air pumping on dissolved oxygen.

great enough, however, as shown by the decline in DO concentration during mixing 4. This occurred in mid-August when temperatures in the lower waters were high and deoxygenation reactions were strong. Early natural fall overturn is shown by the rising DO level at the 0.7 depth in the test lake in mid-September. This did not occur in the unmixed lake.

Figure 7 shows that high concentrations of sulfide in the lower waters of the test lake were avoided by the mixing operation. No sulfides existed in either lake at the 5-foot depth. The behavior of the total iron concentration, shown in Figure 8, is not definitive. No significant difference can be demonstrated between the mixed and the unmixed lakes. Possibly this is because the analytic technique measured total iron, and any soluble reduced iron that was oxidized by the DO provided in the lower waters did not have an opportunity to precipitate below the 0.7 depth. It was included, therefore, in the samples.

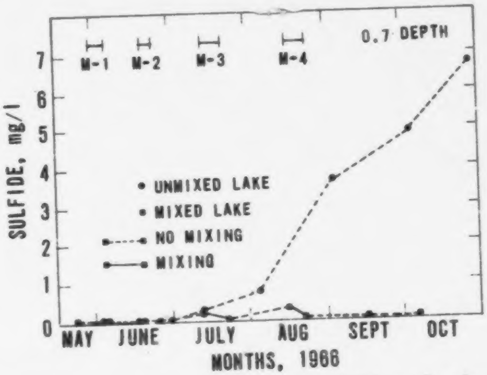

Figure 7. Influence of diffused-air pumping on sulfide.

In contrast, the behavior of the total manganese concentrations in the two lakes, particularly in the lower waters (Figure 9), shows that high concentrations of manganese in the fall season of the year were avoided through the oxidation and precipitation of reduced manganese induced by artificial destratification. Finally, diffused-air pumping had little effect on the concentration of COD in the test lake (Figure 10), possibly because the biodegradable portion of the total COD was small;

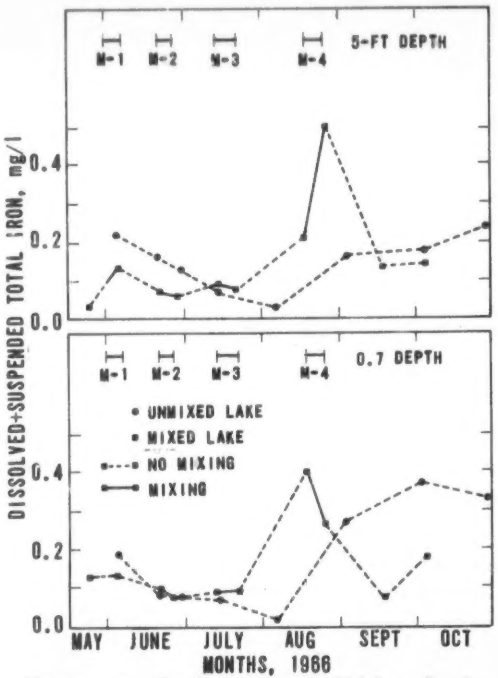

Figure 8. Influence of diffused-air pumping on total iron.

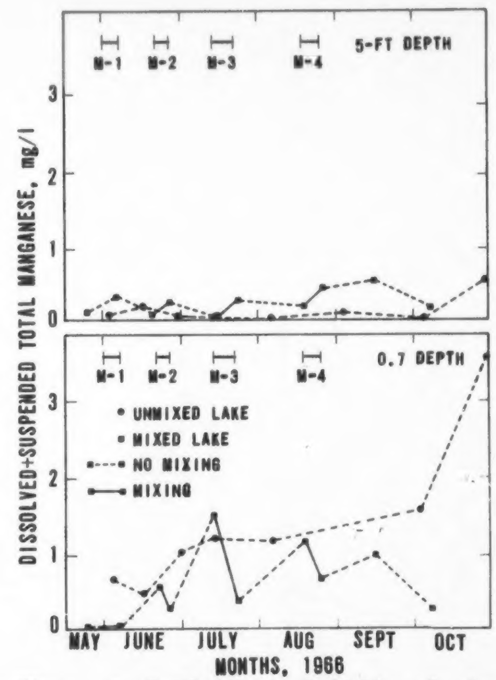

Figure 9. Influence of diffused-air pumping on total manganese.

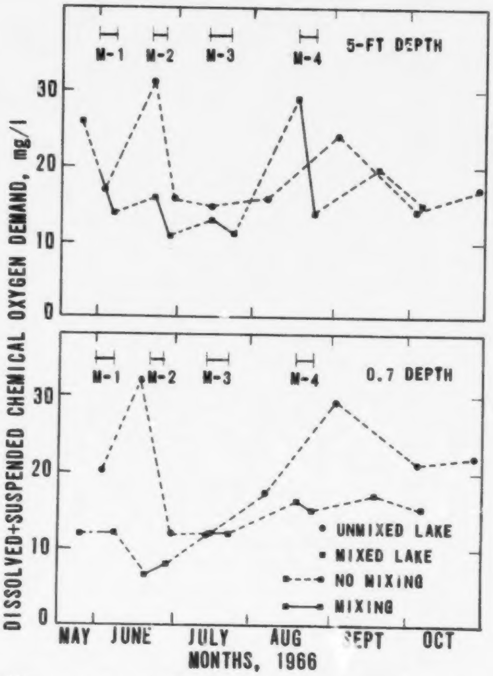

Figure 10. Influence of diffused-air pumping on COD.

therefore, any enhanced biodegradation caused by increased DO concentrations in the lower waters could not be measured. The generally lower concentration of COD in the test lake, as contrasted to the unmixed lake, from mid-July through mid-October was not considered significant.

A previous paper[6] proposed the calculation of an oxygen balance below the 30-foot depth as a method of judging the overall influence of artificial destratification on oxygen sources and sinks in these two lakes. To determine the oxygen balance, as presented in Figure 11, the total pounds of DO, sulfide as the oxygen equivalent, iron as the oxygen equivalent, and manganese as the oxygen equivalent on any given sample day were algebraically summed (plus for DO and minus for oxygen sinks). Because, as shown in Figure 10, the COD was not affected by artificial destratification, it was not included as an oxygen sink. The net oxygen balance in pounds per million gallons for the mixed and the unmixed lakes on each sample day (Figure

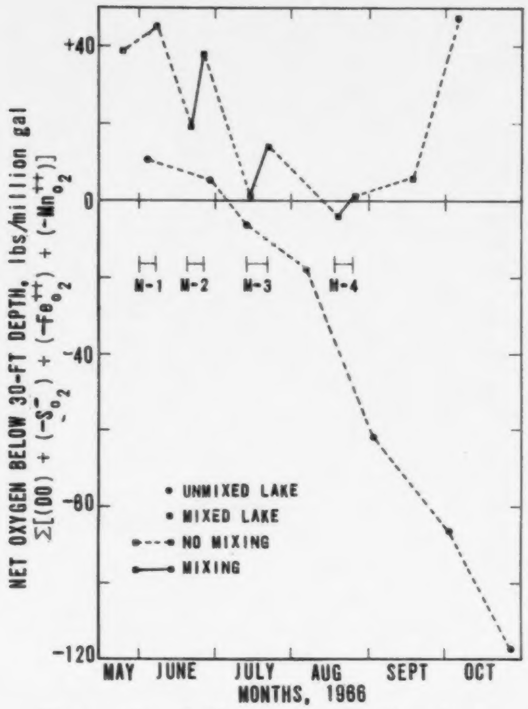

Figure 11. Influence of diffused-air pumping on the oxygen balance below the 30-ft depth.

11) demonstrates the beneficial effect of diffused-air pumping on the test lake. Early fall overturn, promoted by the elimination of a thermal gradient, is shown by the rapid rise in the oxygen resources of the lower waters in the test lake in mid-September. This rise did not occur in the unmixed lake.

Plant Nutrients and Plankton

Figures 12, 13, and 14 show some increases in nitrogen concentrations caused by diffused-air pumping. Nitrite nitrogen data are not presented, as NO_2–N was absent from both lakes. Figure 15 shows only slight effects on the dissolved acid hydrolyzable phosphorus concentrations in the test lake. Even more important than the behavior of the plant nutrients is the resulting effect on the

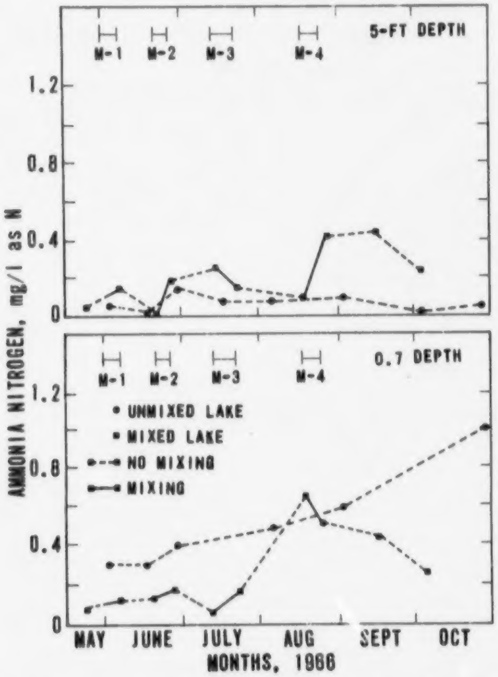

Figure 12. Influence of diffused-air pumping on ammonia nitrogen.

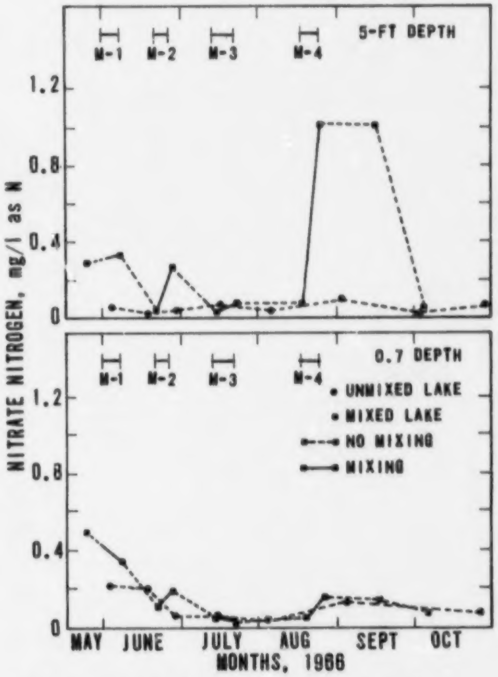

Figure 13. Influence of diffused-air pumping on nitrate nitrogen.

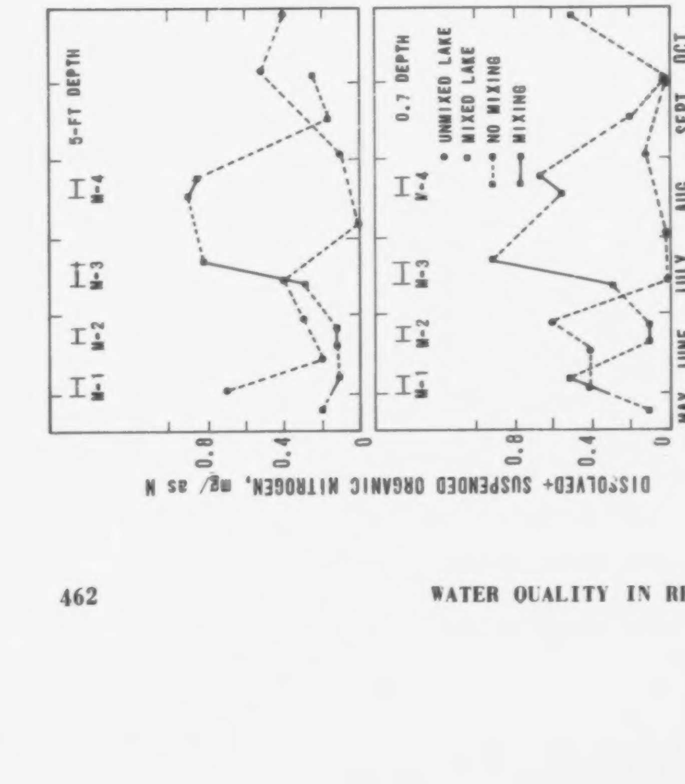

Figure 14. Influence of diffused-air pumping on organic nitrogen.

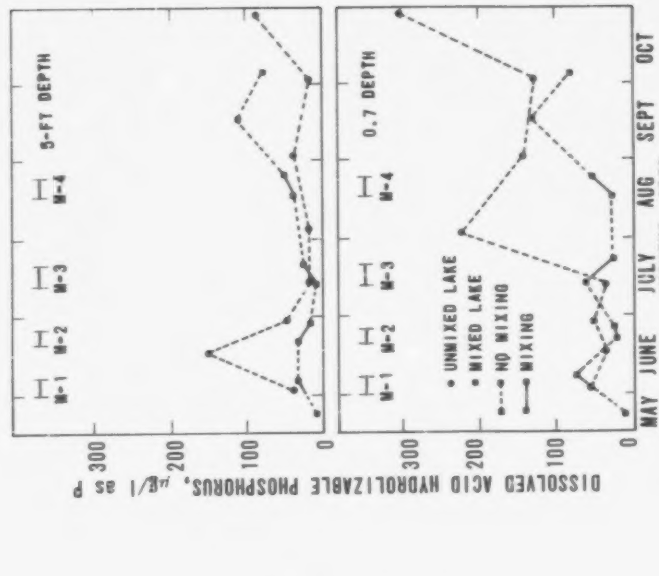

Figure 15. Influence of diffused-air pumping on dissolved phosphorus.

plankton populations. Figure 16 shows that, in spite of some increase in nitrogen concentrations, no sharp increases occurred in total plankton densities either in the upper or the lower waters of the test lake as compared to those in the unmixed lake. Actually, some environmental factor changed during artificial destratification to cause a decline in plankton populations. This decline was only temporary, however, and the plankton densities returned to original levels between mixing periods. The possibility of causing large algal blooms through the process of artificial destratification was considered prior to this study. Algal blooms did not occur, however.

Carbonate Buffer System

Figure 17 shows a temporary decline in pH after each mixing period in the upper waters, whereas little effect

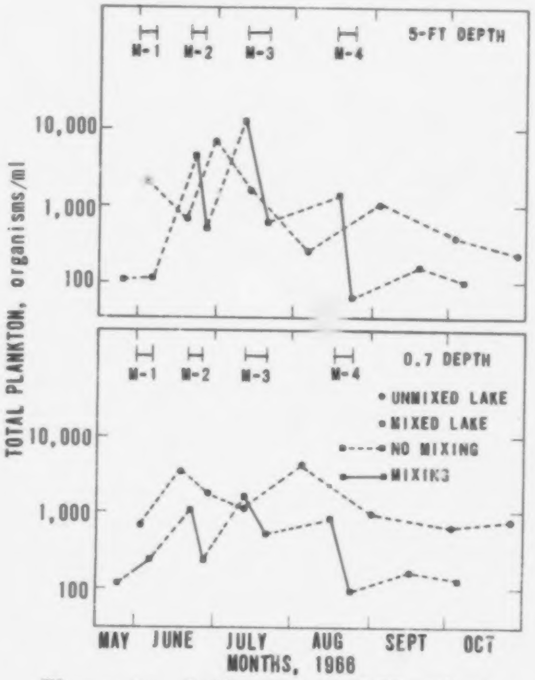

Figure 16. Influence of diffused-air pumping on total plankton.

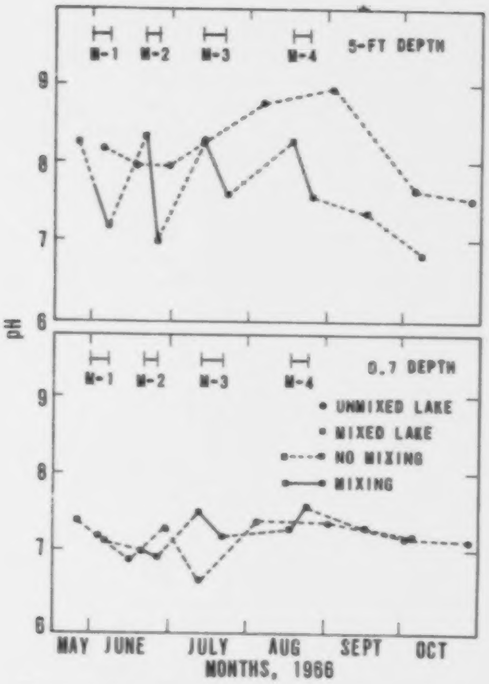

Figure 17. Influence of diffused-air pumping on pH.

is shown in the lower waters. This decline in pH probably was caused by bringing carbon dioxide from near the bottom into the upper waters during mixing. This increase in carbon dioxide concentration is shown in the upper portion of Figure 18. Very little effect on either pH or carbon dioxide concentration was noted in the lower waters of the test lake. Some increase in calcium concentration was detected in the upper waters of the test lake during pumping (Figure 19), but little or no effect was noted at the lower level.

Figures 20 and 21 show that artificial destratification had little or no influence on the concentrations of alkalinity and magnesium in either the upper or the lower waters of the test lake when contrasted with the unmixed lake.

Sulfates, Conductivity, and Particulates

Figures 22 and 23 show that diffused-air pumping had little influence on the concentration of sulfates or

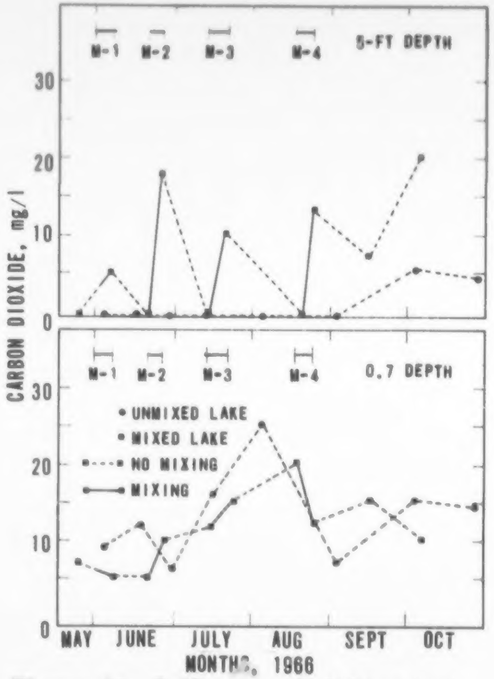

Figure 18. Influence of diffused-air pumping on carbon dioxide.

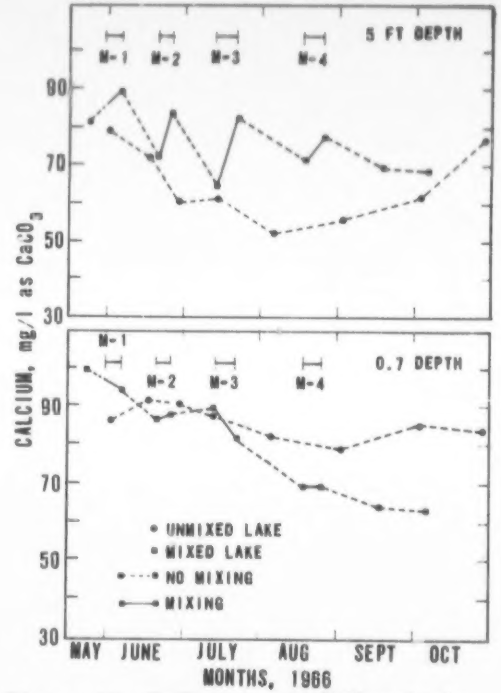

Figure 19. Influence of diffused-air pumping on calcium.

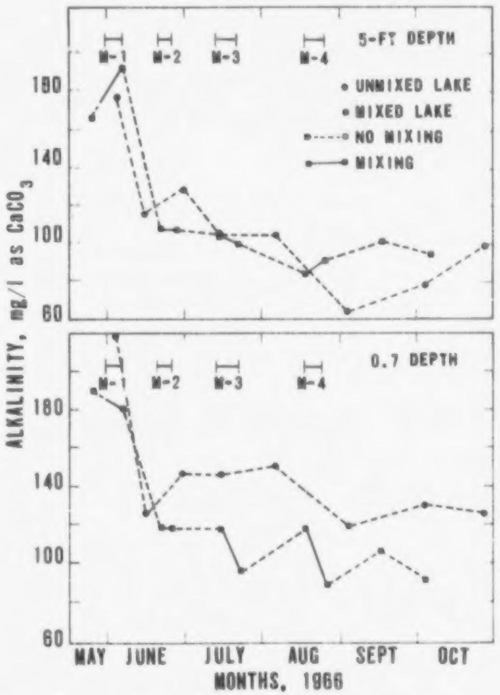

Figure 20. Influence of diffused-air pumping on alkalinity.

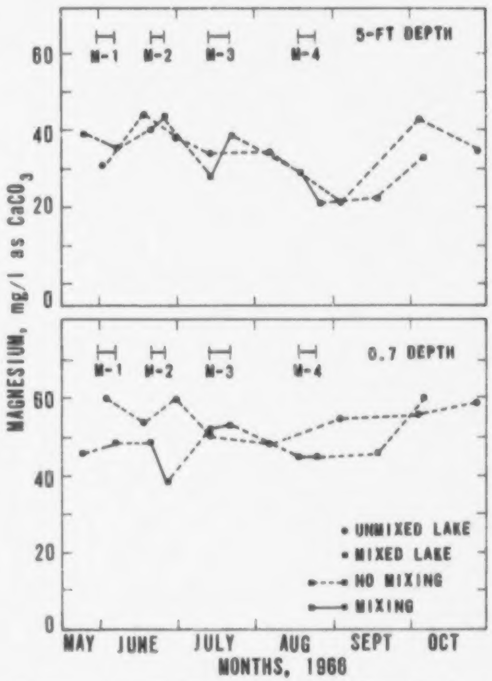

Figure 21. Influence of diffused-air pumping on magnesium.

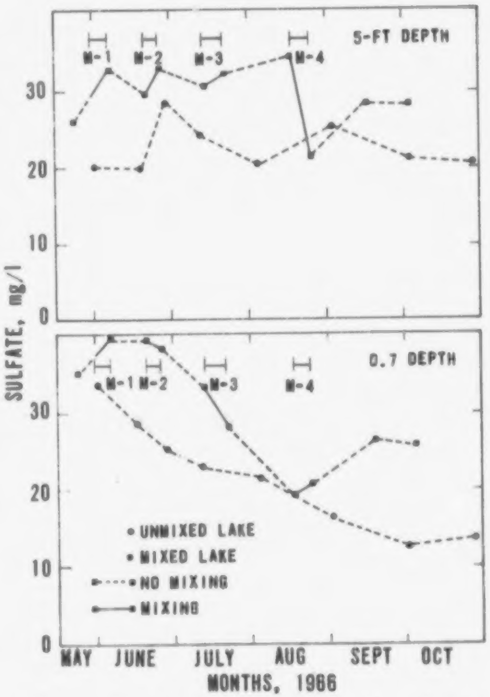

Figure 22. Influence of diffused-air pumping on sulfate.

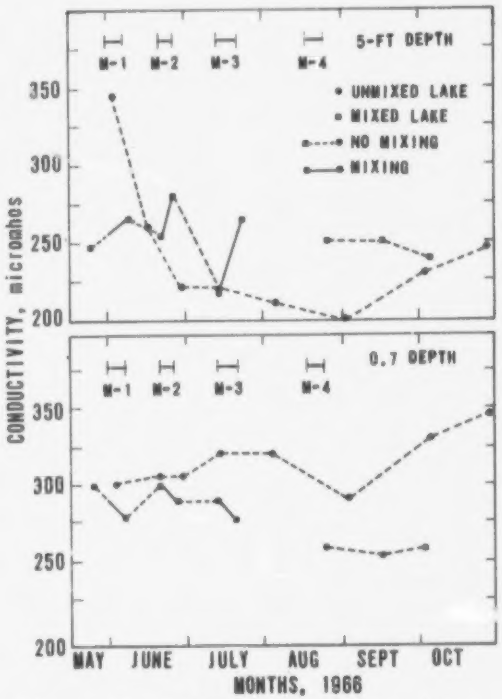

Figure 23. Influence of diffused-air pumping on conductivity.

on the conductivity, as a measure of the total inorganic ion content of the test lake. Although suspended solids concentration or turbidity was not determined during the 1966 experiment, previous studies on the mixing of Boltz Lake with a mechanical pump[2] showed that destratification had little or no influence on the suspended matter in the lake. Also, visual inspection of the rising water in the diffusor area showed the water to be clear, not cloudy. Significant mud was not raised from the bottom.

SUMMARY AND CONCLUSIONS

The data presented in this paper have shown that artificial destratification by diffused-air pumping is an effective method for controlling water quality throughout the entire volume of a reservoir. Many improvements in various water quality parameters were noted, and no deleterious effects could be detected. Future investigations are needed in two areas: other destratification devices and alternate patterns of operation; and the influence of artificial destratification on indicator organisms and pathogens.

From this study, the following conclusions can be made:

1. Artificial destratification will affect only those water quality parameters that; a) initially show a concentration gradient with depth, or b) are influenced by the vertical transport of some other water quality constituent. For example, vertical transport of carbon dioxide will lower the surface pH even if the pH initially did not show a vertical gradient.

2. In this study, artificial destratification affected the vertical distribution of the following parameters, as contrasted with their distribution in a nearby unmixed lake: temperature, concentrations of DO, sulfides, manganese, ammonia nitrogen, nitrate nitrogen, organic nitrogen, plankton (temporarily lowered), hydrogen ions (pH), carbon dioxide, and calcium.

3. Based on this study, artificial destratification did not substantially influence the vertical distribution of the following water quality parameters, as compared with their vertical distribution in a nearby unmixed lake: conductivity and concentrations of iron, COD, dissolved acid hydrolyzable phosphorus, alkalinity, magnesium, and sulfates.

ACKNOWLEDGMENTS

The authors wish to thank Lawrence Kamphake, George Holtzer, and Richard Shibiya for the laboratory chemical analyses, Ernest L. Robinson for the enumeration of the phytoplankton densities, and Glenn R. Gruber for help with the field work. In addition, the cooperation of the personnel of the State of Kentucky Department of Fish and Wildlife Resources is gratefully acknowledged.

APPENDIX

Author's Note: Because the oral presentation at the Research Conference was different from the paper that precedes this Appendix, and the discussion that follows was based on the oral presentation, this Appendix paraphrases the essential points of the oral talk. This paraphrasing was transcribed from the tape recording made at the time of the actual presentation. In addition, some information that developed subsequent to publication has been added. The topics covered during the presentation included: 1) a summary of the water quality improvements created by artificial destratification, 2) a discussion of equipment design based on the ''oxygenation capacity'' method, 3) discussion of equipment design based on the ''destratification efficiency'' method, and 4) a design example of each technique. This transcription is reported generally in oral style.

Before I begin my formal presentation, I would like to put our research in proper perspective. One should select the quality objectives in a given situation and then choose the control technique that will best meet these objectives. This will not be complete mixing in every case, but it will be in some cases. For example, complete mixing would be useful if maintenance of a cool water volume is not required; or a uniform density is required to permit use of the full volume of the reservoir for dilution of inflows or runoff; or if evaporation control is advantageous during mid-summer. Our overall objective since this phase of our project began in 1964 was to demonstrate the effectiveness of this technique, discover any disadvantages, and determine some design criteria, so that anyone considering this method for use would have a firmer basis for judgment.

If we agree that for a particular water quality objective, the technique we desire is complete mixing of a reservoir, the two questions most frequently asked are: 1) what kind of equipment is best, and 2) once the equipment is chosen, how big should it be? Three proprietory products as well as the homemade devices such as the two we used in our studies are available for artificial destratification. These are the types of equipment for which evaluation is desired.

Equipment Evaluation

OXYGENATION CAPACITY METHOD

Because the major influence of artificial destratification is on the oxygen balance in a reservoir, as we have defined it, judgment of various devices should logically be made on the basis of comparing the influence each has on the oxygen balance in a given lake per unit of energy input. To help make this judgment, we have defined a parameter that we call ''oxygenation capacity.'' This is calculated according to Equation (1):

$$\frac{\text{Net change in oxygen balance from } t_1 \text{ to } t_2}{\text{Total energy input from } t_1 \text{ to } t_2} = \text{Oxygenation Capacity (OC)} \qquad (1)$$

where t_1 and t_2 are the times of starting and stopping the mixing.

The units of this parameter are pounds of dissolved oxygen (DO) per kilowatt-hour.

If you could rate various pieces of mechanical equipment on an OC basis, presumably the device with the highest OC would be chosen. Figure 24 shows a rise in the oxygen balance (O_2B) during the mixing period. The OC for the mechanical pump, used in 1965, would be the net ΔO_2B divided by the total number of kilowatt-hours accumulated during the 5-week mixing period. A control was simulated by extrapolation.

When this technique is used to compare various pieces of equipment, four items must be kept in mind, if fair comparisons are to be developed. These are: 1) the OC value is inversely related to the oxygen balance that exists at the start of a given test[6] (the higher the O_2B, the lower the OC); 2) the time rate of change of the oxygen balance curve declines during a given mixing as

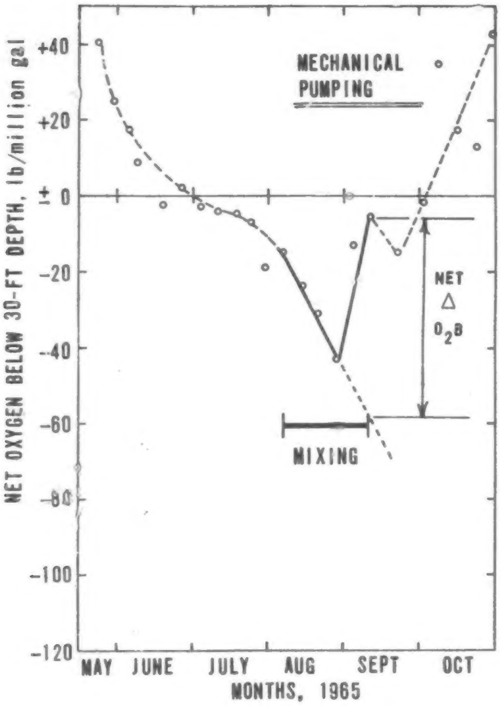

Figure 24. Change in oxygen balance with time.

DO saturation is approached so that the length of a test influences the resulting OC; 3) a control is needed to determine how the oxygen balance would have behaved had there been no mixing; and 4) because meteorological conditions may vary from test to test, comparison should be made under similar meteorological conditions. These difficulties can be overcome, however, and a fair comparison may be obtained among various pieces of mechanical equipment.

DESTRATIFICATION EFFICIENCY METHOD

An alternate approach for evaluating destratification equipment is to use the calculated stability that develops in a stratified body of water. Use of this technique assumes that mixing a lake to remove its stability will permit adequate reaeration of the water. This assumption may be made only if a reservoir is to be completely mixed. Techniques for reaerating the bottom water of a reservoir without upsetting the thermal profile are available.[11]

Because the calculated stability in a reservoir, which may be computed from the temperature profile and a depth-volume curve of the body of water, is the theoretical minimum energy input required to mix the lake, we have defined a parameter related to stability for rating the performance of various pieces of mechanical equipment. This parameter is called destratification efficiency and is calculated according to Equation (2):

$$\left[\left(\frac{\text{Net change in stability from } t_1 \text{ to } t_2}{\text{Total energy input from } t_1 \text{ to } t_2}\right) \times 100\right] = \begin{array}{c}\text{Destrati-}\\ \text{fication}\\ \text{Efficiency (DE)}\end{array} \qquad (2)$$

where t_1 and t_2 are the times of starting and stopping the mixing.

Presumably, the mechanical device with the highest DE would also have the best reaeration characteristics, and would be the best device for use in a given reservoir.

Figure 25 gives an example of the calculation of the DE. In this case, a nearby lake chosen as a control, shows a typical rising and falling stability pattern through the summer season. The test lake, because of somewhat different morphology, has approximately 7 kilowatt-hours more stability than the control lake, at any time before

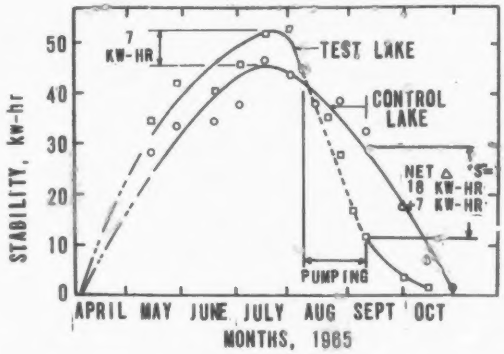

Figure 25. Stability versus time.

the mixing starts. Mixing starts in early August and a rapid decline in stability in the test lake occurs. At the end of the mixing period, the stability of the test lake is 18 kilowatt-hours less than that of the control at the same time. This, plus the 7 kilowatt-hours, which represents the basic difference in stability between the two lakes, yields a total net change in stability of 25 kilowatt-hours during the mixing period. The numerator in Equation (2) is this value, and the denominator is the total energy input during the 5 weeks the mechanical device operated in the test lake. Destratification efficiencies calculated in this manner for various pieces of mechanical equipment could then be compared and the maximum selected.

As in the previous case in which OC was used for rating destratification equipment, difficulties exist when DE is used as a rating parameter. For example, 1) the rate of change of stability with respect to time declines as the mixing operation proceeds. Therefore, deciding when to end a given test will affect the calculated DE.[3]

Also, 2) morphological differences affect the calculation of stability; therefore, comparison of mixing devices in various lakes may lead to faulty conclusions.[3]

> *Author's Note: During the oral presentation, we suggested that stability per unit surface area was not a comparable parameter from lake to lake, but that stability per unit volume might be. During the open discussion that followed, the concept of using stability per unit volume was specifically criticized. Mr. Uttormark pointed out that because the stability of a lake is the vertical distance between the center of gravity when the lake is isothermal and thermally stratified multiplied by the density of water, the product then being multiplied by the volume of the lake, dividing the final product by the volume of the lake, as the suggested parameter stability per unit volume does, merely means that lakes are being compared on the basis of the vertical distances between their isothermal and stratified centers of gravity; he did not think this was very useful.*

Another problem with morphology is one of surface heating. A lake with a large surface area with respect to the total volume will develop stability much faster than a lake with a small surface area with respect to total volume.

This again makes reservoir to reservoir comparisons difficult.[6]

Further, 3) a control is needed to determine what the course of the calculated stability would have been in the body of water under study if there had been no mixing. The *net* change in stability is needed for the numerator of Equation (2) if fair comparisons are to be made.[6]

Finally, 4) meteorological conditions influence stability and destratification, so tests of various pieces of equipment should be made under as near similar weather conditions as possible.

Equipment Design

OXYGENATION CAPACITY METHOD

Although comparison of various pieces of equipment on the basis of short-term OC values probably is valid for choosing equipment, its short-term OC may not be correct for design purposes. An OC based on an entire season, calculated by the "area method" described in Paper No. 12 of this Compilation, should be a more realistic design parameter. The difficulty here, of course, is that the pilot model of the chosen device must be operated such a long time to establish the design OC. Possibly experience will eventually lead to the establishment of general design OC's for different types of artificial destratification equipment. Our experience to date (1968) indicates an OC of 0.7 pound DO per kilowatt-hour for our 1965 mechanical pump mixing experiment and 1.7 pounds DO per kilowatt-hour for our 1966 diffused-air pump mixing experiment, both calculated by the area method.

The following procedure would be suggested if one wished to design a diffused-air pump for one of our study lakes, Boltz Lake. Figure 26 shows the trend of the quantity of DO (DO only; reduced materials are not included here) below the 30-foot depth in a normal year (the solid line). If the DO concentration target to be achieved by artificial destratification were to maintain 4 milligrams or more of DO per liter below the 30-foot depth at all times, the broken line might be the target DO trend. This line shows 4 milligrams of DO per liter through the peak of the heating season and starts to rise in mid-to-late August, when natural mixing because of fall cooling begins. The shaded area between the two curves, 52.6×10^6 pounds DO-hours, is divided by the design OC, 1.7 pounds DO per kilowatt-hour. This gives a number, 30.9×10^6 kilowatt-(hours)^2, that equals the area under the cumula-

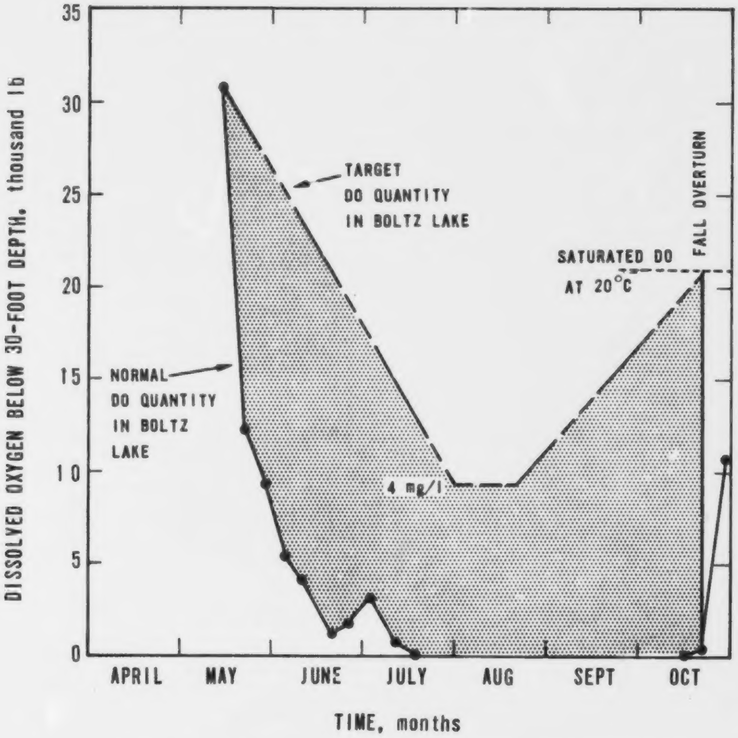

Figure 26. Design data, oxygenation capacity method.

tive energy input versus time curve, hereafter called the ''energy area.''

Because a continuously operating constant energy source produces a triangular shaped energy area, and the slope of the curve is the desired design input power, dividing 30.9 X 10^6 kilowatt-(hours)^2 by (1/2) (t)^2, 7.55 X 10^6(hours)^2, where ''t'' is the operating time in hours, 3,888 hours, yields a design power input of 4.1 kilowatts.

During the 1966 experiment in Boltz Lake, a 22.4 kilowatt compressor was run for periods of 4 to 6 days four times during the summer season. Although the power of this mixing device was five-plus times bigger than the suggested 4.1-kilowatt machine design from the above calculations, the 1966 energy area was only 7.9 X 10^6 kilowatt-(hours)^2. This is only 25 percent of the energy area value suggested above, 30.9×10^6 kilowatt-(hours)^2. Reference to Figure 6 shows that 4 milligrams of DO per liter was not maintained at the 0.7 depth in 1966, however; therefore, the smaller total input of energy was not acceptable from a DO target point of view.

While 30.9×10^6 kilowatt-(hours)^2 may be an acceptable energy area design figure for a continuously operating mixing machine, the assumption cannot be made that any pattern of energy input that produces an energy area of 30.9×10^6 kilowatt-(hours)^2 will produce the same DO concentration trend. Obviously a very large device operating over a short period sometime during a summer season could produce the proper energy area, but it would not produce the desired influence on DO concentration. What is not known at this time (1968) is whether a somewhat larger device operating intermittently each day on off-peak power would be economical, thus trading improved operating cost for a slightly increased capital investment, but still meeting a given DO concentration target. Further research is needed in this important area.

Other problems must also be recognized. For example: 1) an oxygen balance curve developed for one lake may not be transferable to another lake. Where design is required for a lake that has not yet been created, predicting the trend of the oxygen balance curve in this new lake may be difficult. The condition of the bottom and the quality of the water used to fill the lake will influence the shape of the oxygen balance curve. Also, 2) further study may show that the OC value for a certain piece of equipment determined in a given lake for one set of oxygen balance conditions may not apply to another lake. Future work should determine the severity of the above mentioned problems.

DESTRATIFICATION EFFICIENCY METHOD

In a manner similar to that mentioned above, equipment might be evaluated based on short-term DE values, and the equipment with the highest DE might be chosen assuming that effective mixing and effective reaeration are related, but the short-term DE might not be a valid design parameter. As noted in Paper No. 12 of this Compilation, a season-long DE can be calculated by the area method. Once general DE values are developed through experience, they can be used for design purposes. Our experience to date (1968) indicates DE values calculated by the area method of 0.1% for the mechanical pump used in 1965 and 0.5% for the diffused-air pump used in 1966, both in Boltz Lake.

Before discussing how these DE values might be used for design, one might ask, why consider the use of stability as a design parameter at all if the quality target is a certain DO concentration and not a uniform temperature? Two reasons exist: one, stability is fairly easy to calculate requiring only a temperature profile; and two, stability patterns for bodies of water in the same geographical area seem to be sufficiently similar to permit prediction of expected stability in a new lake or reservoir. This similarity might not be true of DO concentration trends. If one does use a stability pattern for design purposes, however, the assumption must be made that complete mixing to remove stability will provide adequate reaeration and meet all oxygen demands.

Figure 27 shows the stability pattern in Boltz Lake in 1965. Experience has shown that, although stability begins to decline in a lake in mid-July because of the lowering of the thermocline below the elevation of the isothermal center of gravity of the lake (see Figure 1 of Paper No. 9 in this Compilation), stability decline caused by fall

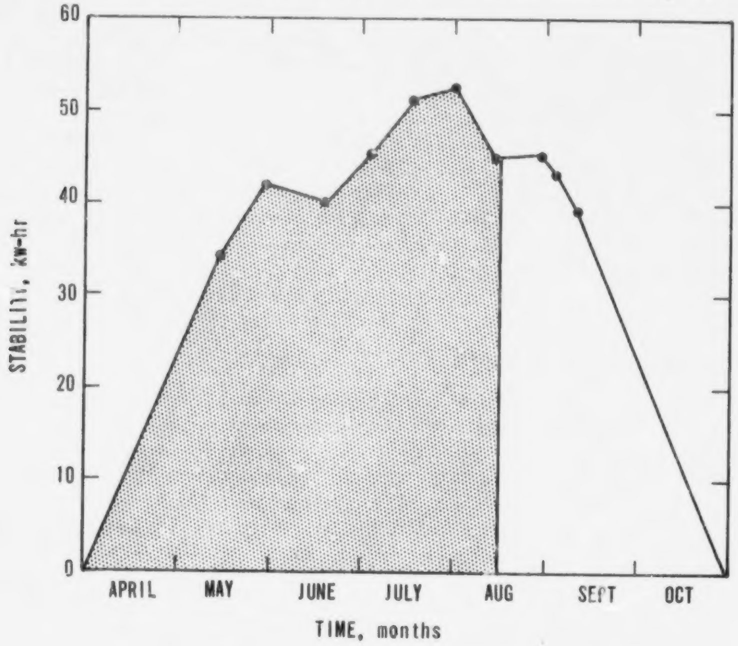

Figure 27. Design data, destratification efficiency method.

cooling starts about mid-August at our latitude of about 38°N. Therefore, our stability target might be to maintain zero stability until mid-August. The shaded area in Figure 27 (114,100 kilowatt-(hours)^2), when divided by the season-long DE (0.005 (0.5%)), yields 22.8 X 10^6 kilowatt-(hours)^2, the energy area for this case. As before, dividing the energy area by $(1/2)(t)^2$, where ''t'' is the time from April first to mid-August in hours (3,288), yields the slope of the energy area curve. This is the design input power that would be adequate if the machine operated continuously. The energy area divided by $(1/2)(t)^2$ yields a slope of 4.2 kilowatts for the design power input.

As noted above, in our Boltz Lake experiment a 22.4-kilowatt compressor operated four times during the summer of 1966 provided an energy area of 7.9 X 10^6 kilowatt-(hours)^2. This is about 35 percent of the required energy area, as calculated above. Reference to Figure 5, however, shows that the stability target was not met. In the 1966 experiment, only 56 percent of the target reduction in stability was made. The reason that 56 percent of the target reduction in stability was accomplished with only 35 percent of the necessary energy area was because stability reduction is more easily accomplished when stabilities are high (see Figure 2, Paper No. 9 of this Compilation).

COMPARISON OF METHODS

The agreement between these two design techniques is probably a coincidence because so many assumptions were made—DO and stability targets, periods of operation, and OC and DE values. In practice, in some biologically active reservoirs additional mixing above that necessary to make the water isothermal may have to be supplied to provide oxygen for the active oxygen sinks. Here, the OC design method would produce a larger result than the DE design method. In cleaner waters DO concentrations might remain adequate, even if some stability remained in the reservoir. In this case, the OC design method would produce a smaller result than the DE design method.

We have not yet had enough experience with these techniques to understand all aspects of the problem. Our judgment at this time (1968), however, is that if an oxygen concentration trend is available from historical records, the OC method of design should be used. Checking this value by the DE method using historical stability records might also provide interesting information.

SUMMARY

In conclusion: 1) we realize some uncertainty presently exists in design criteria, but 2) the advantages of water quality control for certain purposes are so attractive that, 3) this has stimulated several proposals to destratify 100,000- to 300,000-acre-foot reservoirs. We feel certain that with our research and by drawing on the research of some of you in the audience and cooperating in the large field studies that are currently proposed, a firm basis for design and operation of artificial destratification for reservoir water quality control will soon be forthcoming.

Discussion of

"CONTROL OF RESERVOIR WATER QUALITY BY ENGINEERING METHODS"

Edward L. Thackston[*]

The present paper and the preceding ones in the series by the same authors, can be a valuable guide for anyone who is considering using destratification to solve or to help alleviate some water quality problems in a lake or reservoir. The paper graphically illustrates some of the water quality problems that may occur in stratified lakes or reservoirs and what can be accomplished in the way of correction by destratifying it.

In addition, the techniques and concepts developed in the earlier papers on this subject can furnish useful guidelines for selecting and operating the necessary mechanical equipment. As discussed earlier, they do not allow the calculation of absolute answers or the confident prediction of results, and there are many uncertainties and pitfalls for the unwary. It is to the authors' credit, however, that they have been very careful to point out the precautions to be taken, the inadequacies, and the disadvantages of each concept, operation, or group of data. The value of any research is greatly enhanced if its limitations are carefully delineated, for this may prevent inappropriate use of the data or method advocated. Such inappropriate use can easily lead to failure and consequent complete rejection of the idea, even the valuable portions.

The value of the destratification efficiency concept is chiefly in comparing different types of destratification equipment in the research and development phase, so as to determine the most efficient and most economical type.

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The efficiency concept may then be used to estimate the proper size of equipment for a given situation. Once the equipment is installed and operating, however, destratification efficiency should not be an operating criterion. Times and rates of operation should be adjusted with an eye toward achieving the water quality goals. Temperature profiles and stability calculations can be very valuable in showing when stratification is reforming after a mixing, because temperature stratification precedes harmful water quality changes, but there should be no attempt to operate a system to maximize destratification efficiency. Rather, data should be gathered relating water quality to certain mixing patterns and stability patterns on the impoundment in question, and these data used as a guide in achieving the required water quality goals with the least expenditure of energy.

The basic question of greatest importance is, then; ''What is the desired (practical) level of water quality in reservoirs?'' The present paper does not attempt to answer this question, but can be of great value in indicating what water quality changes are possible, and what levels of quality are practical. A great need exists for the definition of realistic, practical water quality goals for reservoirs. The following questions and points will serve to illustrate some of the questions that must be answered, and point out some of the problems involved in answering them.

1. What water quality variables, or parameters, should the goal be applied to? The most obvious choice is probably dissolved oxygen. It is not only important in itself, but its presence, absence, or level controls other parameters of importance, such as dissolved iron and manganese, hydrogen sulfide, and the oxidation state of other constituents. Other variables, however, may need to be monitored and controlled for the effect they may have on DO (examples are BOD and temperature) or because of their importance in themselves.

As with all of these points, no single answer will be applicable to all situations. They will vary with many local conditions.

2. What level of the critical variable should we try to maintain? For the example of DO, is it enough to try to keep the whole reservoir aerobic, with only a trace of DO at all locations being acceptable? This will prevent some problems, but not all of them. Should the level be 2 to 3 milligrams per liter, which will allow fish life to survive, but not thrive? Should the standard apply to

the whole reservoir, or to some fraction of the contents, or to all except a certain layer near the bottom? To what time period should the standard apply? Should it be the whole year, or a particular percentage of the time, or should certain periods have higher or lower standards than the balance of the year? If part of the year is to be treated differently, what part should it be and how long should the period be? The answer to the question of standard levels will, or at least should, depend on the particular harmful effects that may be caused by low oxygen levels in any real situation. This leads directly to point 3.

3. What are the harmful effects of not meeting the standard, both qualitatively and economically? Do low oxygen levels in the hypolimnion affect recreational use of a reservoir, directly or indirectly? If so, what is the effect and how much does it cost? Does the discharge of cold or low-DO water beneficially or detrimentally affect uses downstream such as recreation, fish and wildlife production, waste assimilation, aesthetics (such as hydrogen sulfide odors accompanying zero DO conditions), or domestic or industrial water use?

Hopefully, water quality management decisions will be based on sound economics, with all the associated benefits and costs taken into account. This will require more data on the levels of economic costs and benefits associated with certain water quality changes, especially those associated with recreation and aesthetics.

The location of the harmful effects is also important as it affects the engineering method chosen to remedy them. As an example, consider a case where it is determined that the greatest harmful effects of a stratified impoundment occur because of the discharge of low-DO water through a dam, affecting uses downstream. Could the situation be remedied by overturning just the section of the reservoir immediately behind the dam? The evidence is not conclusive, but preliminary reports suggest this is possible. If so, many harmful effects of very large reservoirs could be alleviated without the necessity of mixing the entire volume. Would preventing the discharge of low-DO water satisfy water quality standards, or must the quality of the entire hypolimnion be raised to meet the standards?

Although the subject paper discussed reservoir mixing as a means of controlling water quality, there are other methods available that should be considered as alternatives in any particular case. The alternative chosen will depend on the characteristics of the local situation, and the type and location of the harmful effects to be alleviated.

Thackston and Speece[12,13] reviewed different methods of mixing impoundments and artificially aerating streams, and Bernhardt[11] described a novel method of aerating the hypolimnion of a stratified impoundment without mixing it. This is useful if it is desirable to preserve the cold temperatures in the hypolimnion for its use as drinking or cooling water.

QUESTIONS FROM THE AUDIENCE AND RESPONSES

R. O. Sylvester, Chairman: Before beginning the open discussion period, Dr. Symons would like to briefly respond to Dr. Thackston's remarks.

Dr. Symons: I would like to thank Dr. Thackston for discussing the water quality aspects of artificial destratification. Our paper emphasizes water quality changes caused by artificial destratification, but because I chose to discuss the design phase of the topic this afternoon, I am glad that someone covered quality.

I would also like to comment on the problem of mixing a lake from a single location. The lakes that we studied were 3 miles long or less. In this situation, we did notice a favorable draining of the cold water from the extremities of the lake to the deepest point where the pumps were located. We could, therefore, mix the entire lake volume from a single location. I have not seen the data to which Dr. Thackston referred, but if in a large lake, 20 or 30 miles long, mixing could only be effected in a 5- or 10-mile zone nearest the mixing device, this may be caused by the low velocities of the cold water moving towards the mixing site. Because only a few summer months are available for mixing, there may be a maximum distance up-reservoir that can be effected from a single mixing location.

The last point I wish to make is in connection with the comparison of mechanical pumps and diffused-air pumps for mixing. Many people are attracted to diffused-air pumping, not because this is basically a more effective way of moving water, but because diffused-air pumps can be more conveniently installed in an existing reservoir after it has been built. The diffusors can be placed from a boat, and the compressor is on shore. This is in contrast to building a rather large permanent structure that would be required if a mechanical pump was to be used for mixing. In a system designed in advance, possibly a large mechanical pump could be provided in the original dam structure; for corrective measures, however, the diffused-air pumping system has seemed attractive.

Chairman Sylvester: Thank you. We will open the meeting to general discussion, and the first discussor will be Dr. Brooks who has a short movie to show.

Norman H. Brooks, California Institute of Technology: Because mixing in reservoirs is really a hydrodynamic problem, we have been seeking to study the hydrodynamics of these systems as another part of our FWPCA-sponsored research. One-half of the problem you have heard about in a previous paper on selective withdrawal, but the other part of the problem is how does water behave if it is discharged into a reservoir? This is what I will show you very briefly in the movie film. Two effects are shown in this film, which shows a jet discharging into a water column; one is the effect of buoyancy, the other, the effect of buoyancy combined with stratification.

> *Author's Note: At this point the movie film showing the discharge of a buoyant jet into a stratified water column, rising to some intermediate level, not the surface, and remaining at that level, was shown.*

John B. Ditmars, California Institute of Technology: In the example you worked for us, you gave us the OC and the DE for the device you used in that case, and you did talk at some length about the numerator in both of these terms, the oxygen balance in one case, and the stability in the other. But you did not mention the denominator, which is the energy that must be put into these systems. You alluded to this problem when you contrasted the mechanical pumping devices to the diffused-air devices, and Dr. Thackston mentioned it when he suggested that possibly diffused-air pumping devices would be more efficient. It is unclear to me why one system is naturally more efficient than the other, just on the basis of the OC and the DE. I was wondering if you could comment on the efficiency of these systems, based on their ability to actually cause more mixing for an equivalent amount of energy; in other words, how much mixing are you getting for the quantity of energy you are putting in?

Also, are the processes the same? When you are mixing with diffused-air pump, in the back of your mind, do you think that you are also transferring oxygen from the bubble to the water, whereas with mechanical pumping you are only moving water around?

Dr. Symons: To answer the last part of your question first, some oxygen transfer occurs from the bubble to the water in the diffused-air system, but we are not counting on this very much. In our opinion, we mainly have a low-head, high-volume air-lift pump in which we are using the diffused air to lighten the water column thereby bringing the bottom water to the surface for reaeration. We have made some attempt to detect any decline in oxygen content of the rising air bubbles. These tests were not too conclusive, but they indicated that if there was any decline in the percentage of oxygen in the air bubble over that of normal air when it reached the surface we could not detect it. Therefore, we do not think we are getting too much transfer of oxygen from the bubble to the water.

To answer the first part of your question, we suggest that the decline in the stability gives an indication of the mixing that is occurring in the reservoir. For the denominator of the equation, we used the rated power of the engine or motor driving our pumping device and multiplied this times the running time to determine the total energy input during any given mixing operation. We have made some attempt to determine the actual quantity of water that we have put in motion with our mixing devices. This is somewhat difficult to do, however, particularly with the diffused-air pump system and therefore we consider the data fairly unreliable.

Mr. Ditmars: Would you care to comment on any comparisons you have between the two systems, mechanical and diffused-air pumping?

Dr. Symons: When we started our mechanical pumping experiments in 1964, we mistakenly purchased a mixed-flow pump instead of a propellor-type pump. We think, therefore, that one reason the mechanical pump produced such a poor DE (about 0.1 percent versus 0.2-0.5 percent for the diffused-air pump, calculated by the area method) was because of a poor choice of equipment. I am somewhat reluctant to report these figures because they seem to indicate that mechanical pumps are not worth considering. We think, however, that a well-designed, low-head, propellor-type pump might rate very favorably, particularly when pumping from great depths.

Paul D. Uttormark, Department of Natural Resources, State of Wisconsin: During the past year I have been associated with a destratification experiment that is being handled by the Wisconsin Conservation Department. This project is somewhat different than the one mentioned by Dr. Symons, because in this case there has been continuous mixing since July 1966. Also, the objectives were somewhat different than those that have been mentioned before. First, I would like to say that this is a small impoundment, surface area of 100 acres and only 28 feet deep. We have installed a commercial product, manufactured by the Aero-Hydraulics Company. This is an air-lift pump and we are pumping from the bottom of the lake at one point. We have six ''guns'' as they are called, which do the pumping.

This project is being operated by the Wisconsin Conservation Department, which looks mainly at the biological aspects. We put the system in the lake, and we tried to obtain continual mixing; then we asked the questions, Would we see a response in the algal population (this lake was very eutrophic prior to operation and exhibited algal scums)? Would we see an increased fish growth, and could we put some bottom organisms into the lake? Prior samples have shown that there are very few bottom organisms in the lake. Prior to this time, this lake had always gone anaerobic in the bottom and samples showed that very few, if any, bottom organisms lived in the lake.

It seems ironic, but in attempting to artificially destratify a lake by pumping from one location, such a system requires some stratification to permit efficient operation. That is, once you have a completely mixed system, the lake does not even know that you have a pump in it. So it is only when you have some stratification that you can bring water from the other portions of the lake to the point of mixing. We found that during the summer of 1967 we created a mixing cell in the lake. We were able to pump water very rapidly in one location, but the rest of the lake was not affected at all.

This mixing cell was very obvious, and could be seen from the shore. At this time, the dissolved oxygen even disappeared from a small portion of the lake and the water was very darkly colored, and you could see it on the surface as a dark area around the mixing devices. The horizontal temperature differences from inside this highly mixed area to just 100 feet away were about 5°F. In the mixing cell, however, we had a very uniform temperature from the surface of the lake to the bottom.

One thing I think this implies for a continuously mixed system is that you will not maintain complete mixing throughout the lake at all times. Some equilibrium condition will develop in which some stability or temperature profile will exist at all times. This is because of the continual input of solar energy that is increasing the stability, and the continual input of mixing energy that is tending to decrease the stability. Under these conditions, an equilibrium situation will develop, but because the efficiency of mixing is somewhat dependent on the degree of stratification of the lake or reservoir, an equilibrium condition will develop in which a lake is not completely mixed.

We saw this in our particular experiment. The hypothesis was that surface temperatures could be decreased significantly by moving cool bottom water up to the surface. We find, however, that when we pump the cool bottom water up to the surface it does not stay at the surface, but mixes and descends to some intermediate depth. We are not, therefore, actually exchanging surface water fast enough to decrease surface temperatures significantly. We have not documented this, however, because in 1967 the summer in Wisconsin was very cool, and therefore the surface temperatures were naturally somewhat cooler than normal.

Some of the other preliminary results showed that, so far, significant increases in fish growth have not occurred, based on average length per age group. The fish involved are bluegills, bass, and northern pike, with the bluegills being studied most extensively. This study will be extended to scale analyses to see if the growth of individual fish was greater since the mixing system was installed.

Finally, I would like to comment on a parameter that Dr. Symons used during his talk. The concept of stability per unit volume has always bothered me, because stability can be defined as the weight of the entire volume of water multiplied by the displacement of the center of gravity from the center of volume. Dividing this by the volume of the lake results merely in the product of the unit weight of water times the distance between the center of gravity and the center of volume. Although the numbers on your chart were expressed in units of watt-hours per acre-foot, so that this was not obvious, this really only measures the displacement of the center of gravity from the center of volume.

Dr. Symons: I think the Wisconsin study is one that we should all watch very closely, and I look forward to its documentation in the technical literature in the near future. Also, I thank you for pointing out the difficulty with the use of stability per unit of volume as a reliable parameter for comparing data from one lake to another. Your objection is, of course, valid.

Edward L. Thackston, Vanderbilt University: I would like to ask Paul Uttormark what happened to the algal populations during the mixing experiment he described?

Mr. Uttormark: We really do not have good algal counts in this experiment. One possible weakness of the experiment is that we are not making algal counts. We are determining the algal populations somewhat indirectly through Secchi Disk readings. We have been taking these data every other day throughout the summer, but it is hard to tell whether or not these readings have been affected by the artificial destratification. There might have been some slight improvement in Secchi Disk readings, but it was not very great. Of course, you must remember, we did not obtain good mixing all summer; at times the mixing cell I previously described was formed, and during those times we did have quite a green lake—I would not call it a scum, but there was definitely a bloom on the lake.

Dr. Thackston: One of the surprising features of the study Dr. Symons reported in the paper that will be printed in the *Proceedings* of this Conference was that algal counts were lowered during the mixing period. Dr. Symons showed a slide during his oral presentation that demonstrated this finding. This finding contradicted a great number of assumptions that had been made prior to mixing studies, and because we do not know exactly why this algal count lowering occurred, this is something that we all should start thinking about.

G. Fred Lee, University of Wisconsin: In your algal counts, could not this lowering simply be accounted for by dilution?

Dr. Symons: We investigated this by calculating the approximate total numbers of algae in the entire lake, both before and after a mixing experience. This was done by converting the algal concentrations at various depths to total numbers by multiplying the counts by the layer volumes. This showed that the total number of algae throughout the lake declined during the mixing period, and that the surface algae were neither diluted by algal-free water nor were the surface populations merely transported to deeper depths. Therefore, we did actually get some die-off of algae in the lake, which we presumed was because of some transient environmental change, maybe cooling, or the change in some key nutrient concentration. But after mixing stopped, the algal populations compensated for this environmental change and grew to near their original levels. Therefore, I do not want anyone to assume that mixing is an algal control technique. Only further research will demonstrate the long-term influence of mixing on algal populations and types.

Dr. Lee: Right. As a second point, one item I think should be included in your oxygen balance computations, which was not included, is the inorganic oxygen demand of the sediment. In Lake Mendota, as an example, 23,000 milligrams per liter of iron in the ferrous form, about 1,000 milligrams per liter of manganese as divalent manganese, and about 2,000 milligrams per liter of sulfide are in the bottom sediment. All this material will exert an oxygen demand of roughly 0.5 moles per kilogram. This will take place quite rapidly when this material is mixed with oxygen. This contribution to the oxygen demand should be included in calculations of lakes of this type.

Dr. Symons: That is a very good point. When an oxygenated layer of water is overlying sediments such as this, how far do you think the oxygen penetrates into the sediment layer?

Dr. Lee: We cannot tell. We do know that the muds are mixed down to a level of 5 to 6 inches each year. This is primarily caused by organisms, however. It is impossible at the present time to estimate the depth of penetration of oxygen or the amount of surface mud in contact with the overlying water.

Dr. Symons: In our case we had a lake with a fairly clean, hard bottom, so not taking this sediment oxygen demand into account in our oxygen balance equation may not have affected our calculations too much. The only problem I see in implementing your suggestion is the difficulty of estimating how much of the inorganic sediment demand to include in the oxygen balance calculation. Would one assume that only a thin layer was oxidized or should the oxygen demand down to some depth into the sediment be included?

Welby R. Madsen, California Department of Water Resources: Do you consider it feasible to install diffused-air pumping systems at a reservoir site where there is no impoundment of water yet, and no experience in operating a reservoir?

Dr. Symons: You would have to make these assumptions: first, that the finished impoundment would thermally stratify, and second, that sufficient oxygen demand would exist in the water and/or the bottom sediment to cause the removal of DO from the water and the buildup of reduced materials. You might be able to judge the truth of these assumptions based on experience with other lakes and reservoirs in the area, and then assume that your new reservoir would behave in a similar manner.

When we first started our study in 1962, we had the idea that we might develop techniques for predicting the behavior of water quality in an impoundment yet to be built, from some knowledge of the characteristics of the soil in the area, typical meteorological information, morphology of the reservoir, and so forth. In the United States, most storage reservoirs will thermally stratify, and this leads to water quality problems. At this time, however, we cannot accurately predict these problems. We are often asked this type of question, particularly if there is a variety of potential reservoir sites under consideration. We have been asked to judge which reservoir site will exert the least detrimental influence on water quality. We have never been able to answer these questions adequately.

This type of problem is now being researched by the Federal Water Pollution Control Administration at the Robert S. Kerr Water Research Center in Ada, Oklahoma. As the pollution control agency, the FWPCA must solve these problems. In the U.S. Public Health Service, our future interest is to investigate artificial destratification as a technique useful to municipal drinking water supplies. Our future work will be concerned with the influence of mixing on indicator organism systems; viral and bacterial pathogens, both in the waters and in the mud; the growth of taste- and odor-causing nuisance organisms; and the impact of multiple-use on water quality. The FWPCA will continue to develop answers to the type of questions that you have raised.

John Anderson, Portland, Oregon: I have reviewed certain reservoirs in the Northwest that have depths of from 400 to 500 feet. My review of the literature showed that most of these artificial destratification schemes have been used in bodies of water less than 100 feet deep. I have had an inquiry at the Aero-Hydraulics Company since September concerning this particular case, to which they have not yet responded. Several systems I have looked at are for destratification in the immediate area of the dam about 1/2 mile upstream rather than the 50 or 60 miles of the whole reservoir. It seems to me that there might well be a problem in reservoirs with these abnormal depths when you are trying just to mix a portion of the reservoir.

Another associated problem has developed in the Northwest, of which I do not think many people are aware. When you destratify using diffused-air pumping, you are adding nitrogen to the water and there is the possibility of making the water supersaturated with respect to nitrogen. Studies made in Vancouver have shown that water supersaturated with nitrogen will kill salmonoid and steelhead fish.

My specific question is this, do you know of any artificial destratification scheme that has destratified a body of water from 100 to 200 feet deep?

Dr. Symons: My search of the literature has shown that 120 feet is about the deepest installation I know of. I have not seen the data from the Ada, Oklahoma, studies so I do not know how deep the Corps of Engineers reservoir was that they destratified. [Eufaula Reservoir is 73 feet deep.]

I would also like to comment on your remark suggesting the possibility of destratifying only a portion of a large reservoir. I have often heard this suggestion made, and it is appealing, as the volumes involved are small with respect to the total volume of the entire reservoir. In my opinion this cannot be done if a system such as I have been discussing is used. Treatment of the discharge water to raise its DO content, thereby preventing downstream problems, may be possible, but if moving of the high density water to the surface for reaeration is the quality control method of choice, this cannot be done in a restricted zone. As high density water is removed from the hypolimnion, more will flow to the pumping location, and a large zone of the reservoir will be influenced.

Dr. Brooks: I would like to add a comment about the problem of mixing deep reservoirs. In deep reservoirs there may be a relative advantage in mechanical pumps over diffused-air pumps for mixing. Using compressed air, at deep depths, one has to raise the air to a very high pressure to overcome the hydrostatic head and get the air to flow out of the diffusor system. To pump water up from depths of 400 feet, however, very little additional head is required because of the hydrostatics of the situation.

Dr. Symons: I am glad you brought that up; this is a valuable point to remember when considering artificial destratification in deep reservoirs.

Donald R. W. Harleman, Massachusetts Institute of Technology: In the denominator of your destratification efficiency equation, you used the total energy input to the compressor. This energy input, however, is a function of the pressure of the air and the volume rate of flow. Therefore, you could put in the same energy with high pressure air at a relatively low volume rate of flow, or low pressure air at a relatively high volume rate of flow, and these two techniques might not have the same effectiveness for mixing. I wonder whether or not you took into account the fact that total energy input to the compressor is not necessarily a measure of the mixing ability of a given system?

Dr. Symons: We did not attempt to optimize the mixing for a given energy input by varying the air pressure and air flow rate. This was beyond the scope of this phase of the project.

Your question brings out another useful point, however. As this project was originally conceived, it was to be staffed with professionals having a variety of disciplines. Limitations of budget, however, prevented us from securing all the staff we desired. Because we had no hydrodynamicist on our team, we did not choose to emphasize this phase of the problem. This admittedly has been a weakness of our project, but I believe our strength is that we did call attention to artificial destratification as a water quality control technique, through our field studies. We intend to continue work in this area, but we hope that university researchers like yourself and others will become sufficiently interested in the potential of this process to apply your knowledge and help optimize this technique.

Dr. Thackston: In connection with Dr. Brooks' previous remark that compressing air to high pressures for release at great depths in reservoirs could be very expensive, I would like to make this comment. One of the earliest trials for artificial destratifying a small water supply reservoir was done by Riddick in New York. In this case, he did not put his air in at the bottom of the reservoir; he put it in right above the thermocline. This set up a rolling motion in the epilimnion, and the shear of the epilimnion rolling across the top of the hypolimnion eventually mixed the entire volume of the reservoir.

His data indicated that once his lake was mixed, putting air in at the 1/3 depth (I believe that was 8 feet) brought water all the way from the bottom to the surface. In a very deep reservoir, therefore, putting air in right below the thermocline may set up inertial currents that would bring water from the bottom of the reservoir, or at least from some depth well below the depth of the diffusor, to the surface. Through the use of this technique, one might be able to save a great deal of money, because the air would not have to be compressed too much.

> *Author's Note: The following statement that pertains to the above discussion was added during transcription.*

Dr. Symons: Riddick's reservoir had a capacity of 103 million gallons (33 ac-ft) and the temperature data showed that an 8-horsepower (6-kw) compressor ran continuously for 7 days to make the water isothermal. This is an energy input per unit volume for one mixing of

$$\frac{1005 \text{ kw-hrs}}{33 \text{ ac-ft}} = 30+ \frac{\text{kw-hrs}}{\text{ac-ft}}$$

Francis C. McMichael, Mellon Institute: I have some general comments that follow from Dr. Harleman's remarks. There is a whole field of literature in the chemical engineering journals that involves the scaling of mixing devices for various industrial processes. Normally, the requirements here are different from those encountered in environmental systems. Generally, in industrial processes, the objective is to obtain a rapid mixing in a relatively short period of time; the time scales are measured in seconds and minutes rather than in days and months. But the process of scaling from one size unit to another still exists. The limitation on the scaling in these cases is, however, generally only up to tanks that are about 150 feet in diameter, which is still small compared to a lake.

Nevertheless, this process of scaling up has lead to many arguments over what is the proper scaling procedure. Many of these scaling techniques have involved some parameter like power per unit volume or some type of energy per unit volume as a measure of the efficiency or performance of a particular size unit. This then is used as a criterion for scale-up. When considering power, however, we must remember that we can put the same power into a system with a small propellor running at a high speed or a large propellor running at a fairly slow speed. Some experience indicates that when you are talking about propellor devices, the most successful mixing devices are those in which the scale of the propellor is nearly the same as that of the vessel. This, of course, is somewhat impractical in environmental systems; we really cannot think of large diameter propellors in this case.

Others in the literature, however, tend to disagree with this concept of power per unit volume as the scaling criterion. This group tends to lean towards momentum per unit volume rather than power per unit volume as the scaling criterion. This again tends to indicate that efficient mixing devices are ones that move relatively large volumes of water at relatively low velocities rather than those that move small volumes of water at relatively high velocities. Although this may not be directly related to the problem of mixing of lakes and reservoirs, I do think that some benefit could be gained from looking at some of this chemical engineering literature.

Dr. Symons: I appreciate your comments. This is the type of information I hope eventually will be brought into the analysis of this problem.

Chairman Sylvester: With that comment I declare the ASCE Specialty Conference adjourned.

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Orlob, G. T., and C. W. Chen. Destratification evaluation techniques.

Symons, J. M., J. K. Carswell, and G. G. Robeck. Mixing water supply reservoirs for quality control.

Knoppert, P. L. Destratification experiments at Rotterdam.

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PART IV

SUMMARIZING PAPERS

PAPER NO. 15

IMPOUNDMENT RESEARCH: KEY TO STREAMFLOW REGULATION PROBLEMS

J.M. Symons and G.G. Robeck

This paper, prepared at the request of the Editor of Water and Wastes Engineering, is a condensed version of Paper No. 2. It contains the historical background of the project and some early research results. Because it was published before the formation of the Federal Water Pollution Control Administration and its subsequent transfer to the U.S. Department of the Interior, some of the organization designations contained in the paper are no longer accurate.

The correct reference citation is:

Symons, J. M. and G. G. Robeck. "Impoundment Research: Key to Streamflow Regulation Problems." Water and Wastes Engineering, 3, Part I, No. 1, 42-44. January 1966; Part II, No. 2, 66-68. February 1966.

INTRODUCTION

Streamflow regulation, as currently viewed, is a method of water quality control in which good-quality stored water is added to a stream at times of deteriorating stream water quality.

Both Federal legislation and practical considerations as to the number of existing reservoirs, and of the number scheduled for construction, dictate the necessity for study of streamflow regulation and its possible applications.

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Based on a review of existing literature[1] on this subject and on the experimental research studies reported here, the authors conclude that:

The need for additional research on the pollution and behavior aspects of both flowing and impounded waters is vital. The fact becomes obvious when one considers that the Corps of Engineers alone currently has over 200 new construction projects authorized and that, for each of these, consideration must be given to the inclusion of storage capacity for water quality control. (The Secretary of the Department of Health, Education, and Welfare will be called on to give advice as to the potential benefit of each of these projects.) While evaluations currently are being made, benefits can be improved only through further research.

Even if water quality control is demonstrated to be an uneconomical use of impounded waters, the information gained through research will still have important value in understanding the behavior of the several hundred impoundments in the United States, many of which are used for water supply.

The trend is toward the complete control of the flow of major rivers. Many rivers, such as the Missouri, are merely a series of impoundments from headwaters to mouth. Since the number of rivers controlled in this manner will increase, it becomes increasingly important to understand impoundment behavior if quality water is to be maintained in these controlled situations.

The research currently being carried out by the Engineering Research Section at the Robert A. Taft Sanitary Engineering Center will be expanded into the new water laboratories in the field when they are activated. This expanded research program will supply answers that are needed to put streamflow regulation projects on a firm scientific foundation.

This article examines the subject by answering four questions frequently asked and by presenting the overall objectives of the program and some of the research findings. The paper thus serves as an early progress report on this long range project. The first part answers these four questions:

What is streamflow regulation?

Why is the U.S. Public Health Service involved?

Why is research needed?

What approach is being taken?

Part one also describes the experimental equipment used in the studies reported and includes some general data. Later detailed data and observations with respect to these simulated impoundment studies will be presented.

Background

About 2½ years ago the U.S. Public Health Service embarked on a research program designed to further the understanding of the behavior of streams and impoundments. The group undertaking this research has been assigned to a project entitled "Effects of Streamflow Regulation on Water Quality." This project is currently being researched at the Robert A. Taft Sanitary Engineering Center in Cincinnati, Ohio. Initial efforts have been to define the scope of the problems by reviewing the literature in order to summarize what is known on this subject and draft a statement of major research needs. In addition, laboratory, pilot, and small-scale field research has begun on some of the aspects of impoundment behavior.

What is Streamflow Regulation?

Streamflow regulation, as defined above, is a method of water quality control in which good-quality, stored water is added to a stream at times of deteriorating stream water quality. In practice, there are three parts to a streamflow regulation scheme: 1) the deterioration of downstream water quality must be determinable; 2) good-quality, impounded water must be available for release and must be released; and 3) there must be a beneficial effect downstream from the release of this impounded water. These three phases of streamflow regulation projects lead to the division of the overall research effort into three parts: "Reevaluation of Pollutional Criteria," "Downstream Effects of Impoundment Releases," and "Influence of Impoundments on Water Quality".

Why is the PHS Involved?

The involvement of the U.S. Department of Health, Education, and Welfare, and ultimately the Public Health Service, began formally on July 20, 1961, with the enactment of Public Law 87-88. This act amended and added several significant paragraphs to the Federal Water Pollution Control Act of 1956, Public Law 660. The pertinent sections (paraphrased) state that the Secretary of the Department of Health, Education, and Welfare shall conduct whatever basic and applied research is necessary to permit evaluation of any streamflow regulation project for water quality control proposed by a Federal construction agency and that he must be ready to supply advice on such projects to the construction agencies.

Although this research is being carried out under Federal mandate, the information to be derived from this project is potentially so important and is needed by so many public and private groups that research in this area would have been initiated even in the absence of Federal legislation.

Why is Research Needed?

Casual observation of the three areas in which research is intended might lead one to question whether this research is needed since research, including pollutional aspects, has been done on the behavior of natural waters for years by sanitary engineers and limnologists. While this past research is most helpful and lays a firm foundation for present efforts, new questions are being asked and, therefore, new answers must be determined.

For example, the traditional BOD test and the DO concentration in a stream are no longer adequate criteria for judging the presence or absence of pollution. A new group of pollutants are factors today: synthetic toxicants, e.g., pesticides; industrial organics, e.g., compounds isolated from CCE material (carbon-chloroform extract); fertilizing nutrients, e.g., nitrogen, phosphorus, and trace elements; dissolved salts; soluble iron and manganese; and heat. Research must develop methods for determining these pollutants and must establish acceptable levels for them. In addition, traditional pollutional criteria must be re-examined in the light of up-to-date knowledge. For example, the dissolved oxygen budget in streams should be reconsidered and attention given to the influence of algal action.

Additional research is also needed on the effects of impoundment releases on water quality downstream. When additional water is added to a stream, many changes take place. Turbulence may be increased, thereby, increasing reaeration; but depth also is increased, which tends to reduce reaeration. Also, flow time between two points is decreased; therefore, the time available for a specific organic to biodegrade is reduced. In some situations the concentrations of organics downstream may be higher after dilution than before. Finally, the effectiveness of releasing dilution water may be limited because the water may hug one shore and not become mixed throughout the entire body of the stream. Only through continued research into these and other related topics can there be assurance that the addition of impounded dilution water at times of pollutional stress will actually have the desired beneficial effects.

Most of the initial effort of the project has been directed toward the remaining phase of streamflow regulation projects, that is, the influence of impoundments on water quality. The environmental conditions in an impoundment differ from those in a flowing stream in many ways; for example, increased detention time and the possibility of thermal stratification. Each condition has an influence on the natural biological, physical, and chemical processes that cause changes in water quality. Many of these processes have been studied under flowing-stream conditions but have not been investigated in impoundments.

The purpose of the pertinent sections of Public Law 660, as amended, was to establish streamflow regulation systems that would provide for impounding water during times of relatively high streamflow and releasing this impounded water at some subsequent date. If such a system is to be successful, the quality of the impounded water must be predictable. If the quality of the impounded water is potentially poor, the ability must be developed to manage the impoundment in such a way as to improve the quality to an acceptable condition.

With specific technical information and the ability to alter the quality, impoundment releases will be of better quality than that of the stream into which they are released. This is necessary if a benefit is to be gained. At present, the influences of an impoundment environment on water quality changes are relatively uncertain. Therefore, if water were impounded, it would not be possible to state with certainty what the quality of this water would be if it were released at some time in the future for the purpose of streamflow regulation. Predictions of impoundment and stream behavior, however, are being made with the best information available.

Considering the large number of writings on the general topics of stream and impoundment behavior, it may be surprising that this uncertainty still exists. There are at least two reasons.

First, much of the data, though generally applicable to the understanding of impoundment behavior, was not taken with impoundment behavior specifically in the researcher's mind. For example, extensive research by limnologists and aquatic biologists has related the growth of algae to light intensity and penetration. Little attention, however, has been paid to the increased light penetration that is caused by impounding a turbid river water, and to the potential problem of organism growth that this clarification may pose.

Second, many of the investigations have not been fundamental enough to allow the phenomena observed to be projected into a new set of environmental conditions. For example, for years, thermal stratification has been noted in impoundments, yet, the necessary set of environmental circumstances that enhance the possibility of thermal stratification has never been completely delineated.

In general, the research efforts on the influence of impoundments must be directed toward 1) thorough investigation of new relationships and 2) correlation of old relationships with the environmental factor, or factors, that causes them so that, in a different environment, different results could be predicted.

What Approach Is Being Taken?

Of the many research avenues considered and discussed by the project staff, three are sufficiently advanced to permit reporting at this time. Two are bench- and pilot-scale projects concerning the behavior of water quality in impoundments, and the third is a small-scale field project concerning a management technique for the maintenance of good quality water in an impoundment.

At this time, a difference of opinion exists as to the selection of the best method of developing techniques for forecasting water quality. One method would measure the changes of a particular water-quality characteristic in several different environments. Then, by relating the rate and extent of the measured change to the total environment in which the change took place, an all-inclusive predictive scheme could be developed to relate environment to change in characteristic. In a new situation, one would need only to measure the essential environmental factors, consult the developed theory, and predict the change in this water-quality characteristic for this new environment. This is an ideal situation because, if the theory, once developed, were truly all-inclusive, the particular characteristic henceforth would be completely predictable. The weakness of this reasoning lies in the development of truly all-inclusive theories. Such theories may be beyond the scope of present ability, since so many factors in a natural environment exert subtle influences on changes in water quality. Currently, however, the extent of this problem is not known.

An alternative approach is the development of a series of operational tests in which the water-quality-characteristic behavior would be measured under a controlled environment sufficiently similar to natural conditions to ensure that the data would be applicable to the field situation. This is what the 5-day BOD test tries to do for the oxygen budget of a flowing stream, although the bottle environment is often different enough from natural conditions to make the data inapplicable in the field. Although this approach would require several tests for each new impoundment, it might be meaningful if a relatively simple set of operational tests could be developed.

At this time, both avenues are being explored. In the current research efforts, the environment is sufficiently controlled to permit the observed results to be related to pertinent environmental factors. Simultaneously, testing procedures are being designed to provide a general relationship between environment and results. If this relationship cannot be developed, the operational tests will be sufficiently realistic to ensure that the data taken will be applicable to a field situation. It is hoped, therefore, that these studies will result in either useful theories, useful operational tests, or both.

INITIAL RESEARCH EFFORTS

The literature in the area of streamflow regulation for water quality control is quite extensive. A comprehensive review, entitled ''INFLUENCE OF IMPOUNDMENTS ON WATER QUALITY – A Review of Literature and Statement of Research Needs,''[1] contains more than 140 references. It was prepared as a part of this project and is available in the Environmental Health Series of the U.S. Public Health Service. Additional information concerning the project staff's research efforts are contained in an article by Symons, Weibel, and Robeck.[2] An overall view of present knowledge and thinking in this area may be obtained from References 3 through 16.

Small-Scale Impoundment Simulation

Fundamental studies on the behavior of water quality in impoundments require an apparatus that allows measurements to be made under controlled conditions. Although any data taken on a small-scale apparatus must be checked eventually under field conditions, the natural environment is so uncontrolled that fundamental studies are difficult.

Figure 1 shows a cross section of the 500-gallon tank developed to simulate a stratified impoundment. By cooling the lower portion of the water column, heating the upper portion, and insulating the sides of the metal tank, both thermal and chemical stratification was accomplished. Upper-stratum circulation and reaeration is accomplished by blowing moist air (moist to reduce evaporation) over the surface of the liquid and by continually stirring the surface of the water with a small laboratory mixer. Six sampling ports allow withdrawal of a sample from several levels.

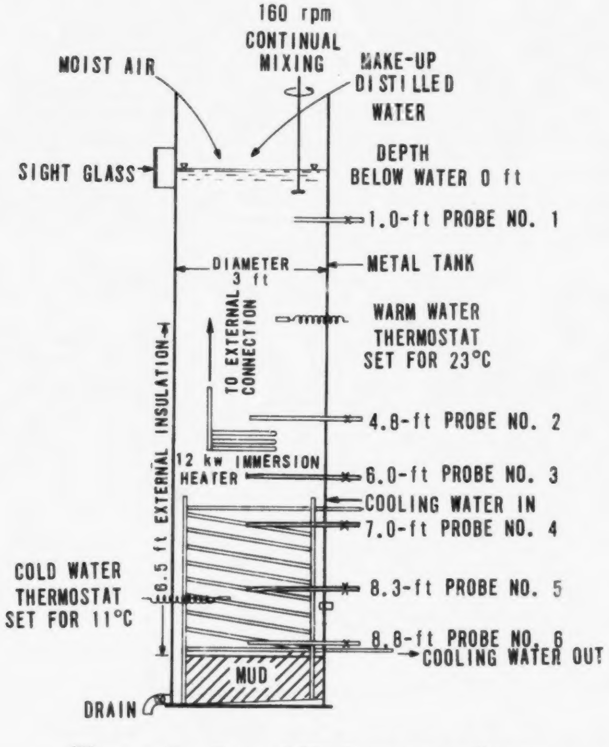
160 rpm
CONTINUAL MIXING
MOIST AIR
MAKE-UP DISTILLED WATER
SIGHT GLASS
DEPTH BELOW WATER 0 ft
1.0-ft PROBE NO. 1
DIAMETER 3 ft
METAL TANK
TO EXTERNAL CONNECTION
WARM WATER THERMOSTAT SET FOR 23°C
6.5 ft EXTERNAL INSULATION
4.8-ft PROBE NO. 2
12 kw IMMERSION HEATER
6.0-ft PROBE NO. 3
COOLING WATER IN
7.0-ft PROBE NO. 4
COLD WATER THERMOSTAT SET FOR 11°C
8.3-ft PROBE NO. 5
8.8-ft PROBE NO. 6
COOLING WATER OUT
MUD
DRAIN

Figure 1. Stratified water column with 500-gallon capacity

Figures 2 and 3 show the extent of thermal and chemical stratification in the simulated impoundment. Figure 2 shows a very sharp temperature gradient between the 6- and 7-foot depth. The other data show: 1) the unit is chemically stratified, 2) the DO concentration is practically zero in the hypolimnion, 3) the epilimnion has a DO concentration nearly 90 percent of saturation, and 4) relatively high concentrations of iron and manganese have been leached from the mud in the anaerobic bottom zone.

The nitrogen data in Figure 3 show that the ammonia nitrogen concentration is higher in the bottom waters than in the top waters. The reason for this difference

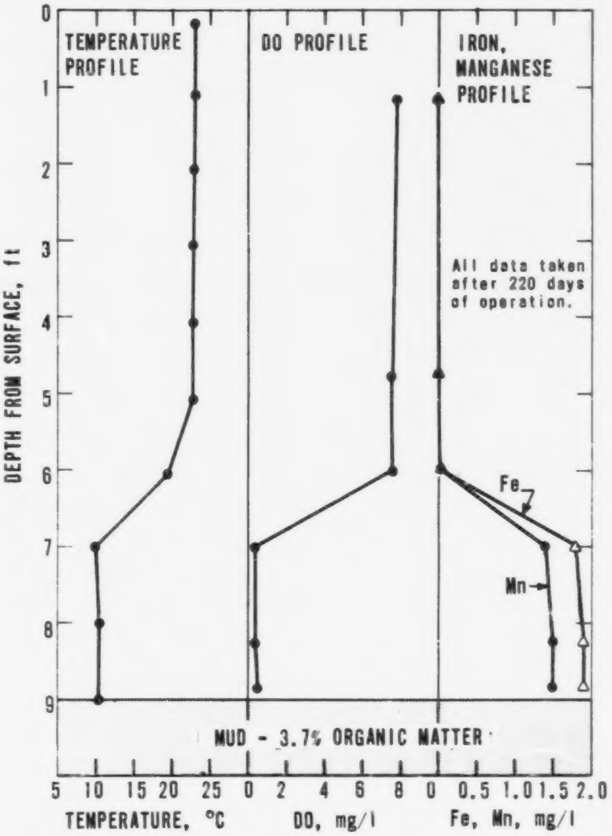

Figure 2. Data from simulated stratified impoundment.

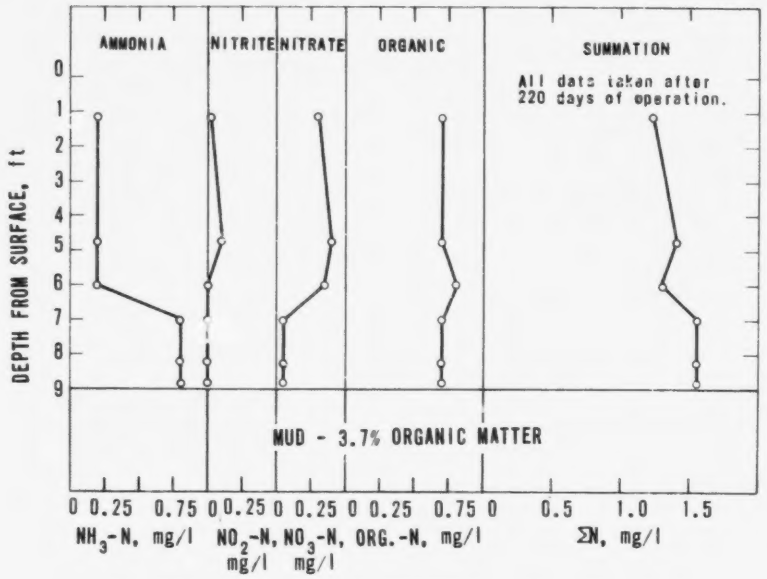

Figure 3. Data on nitrogen relationships in simulated stratified impoundment.

may be the ammonia which is produced during the anaerobic degradation of proteinaceous material that may have originally been in the mud, or which settled into the bottom waters. Figure 3 also shows that, as expected, there is an absence of oxidized nitrogen forms in the anaerobic bottom waters. The discontinuity in the ''ΣN'' curve at the 6-foot level is probably caused by analytical error. Tracer data showed the top waters to be mixed every 20 minutes, but little exchange between the top and bottom waters. It is evident from all of these data that this device behaves in a manner similar to that of a stratified impoundment in the field. This useful apparatus, developed for the study of the behavior of water quality in a stratified impoundment, is currently being used to compare the rates of biodegradation of organic matter in both the upper and lower waters of a stratified impoundment.

The general deterioration of water quality in the hypolimnion of a stratified impoundment has been recognized by many investigators.[8,9,13,14,15] If sufficient energy could be added to overcome the forces of stratification caused by the difference in density of water at different temperatures, no anaerobic zone could develop, and the water quality in the impoundment should improve.

Artificial Destratification

Since 1952, when the first reported attempt at artificial destratification was made by Hooper, Ball, and Tanner[17] sixteen other attempts to mix impoundments have been reported. All the experiments except one were successful, and the investigators reported the presence of oxygen in the lower stratum of water. All the attempts to date have been made on relatively small bodies of water, (maximum, 40,000 acre-feet), so that the practicality of mixing a large storage impoundment has yet to be demonstrated.

In a stratified impoundment, the thermocline is defined as the zone of water in which the temperature gradient is equal to or greater than a decline of 1°C per meter of depth. The generalized theory is that the water above the thermocline is mixed by the wind, while the water below the thermocline is stagnant. Since, however, the decrease in density for each degree C is much greater at warm temperatures than at cold ones, even if an impoundment showed a rather modest temperature gradient, never reaching the intensity of a thermocline, it still could be density stratified. The density profile, not the temperature profile, therefore, determines the thickness of the stratum of water that must be mechanically mixed if artificial circulation is to be accomplished.

Since the volume-depth curve of most impoundments is not linear, a rather small volume of water relative to the overall volume of the impoundment will be contained in the lower stratum that must be raised to surface, i.e., a relatively small volume of water may actually have to be moved. In a lake in the vicinity of Cincinnati, Ohio, only about 20 percent of the volume lies below the mid-depth.

To determine the feasibility of artificially mixing a stratified impoundment, an attempt was made to destratify Vesuvius Lake near Ironton, Ohio in September 1964. Vesuvius Lake has a volume of 1,260 acre-feet, a surface area of 100 acres, and is approximately 3 miles long with a maximum width of 800 feet. The mixing device was a 2,880-gpm (13 acre-feet per day) mixed-flow pump driven by a 16-horsepower gasoline engine. The pump, which was mounted on a raft positioned over the deepest part of the lake, drew water from just above the bottom and discharged it at the surface. The pump was operated for about 200 hours in a semicontinuous fashion during a 10-day period.

The results of this operation are shown in Figure 4. The temperature data show that the impoundment was mixed. In addition to creating a uniform temperature at the pumping site, temperature data taken at the extremities of the lake showed that the surface temperature of the entire lake was lowered from 27.5 to 21°C. From the temperature data, it would appear that the entire lake was well mixed. The dissolved oxygen data are not quite so clear, however. While the dissolved oxygen concentration in the surface was decreased and in the lower waters increased, as would be expected from mixing, the lack of uniformity in the dissolved oxygen concentration throughout the depth at the pumping site was not explainable at the time. Whether or not this means that, contrary to the temperature data implications, the entire lake was not completely mixed, can only be decided after further research. The initial attempt was sufficiently successful, however, to encourage the project staff to pursue further this type of operation. In the future, attempts at artificial destratification with this and other types of

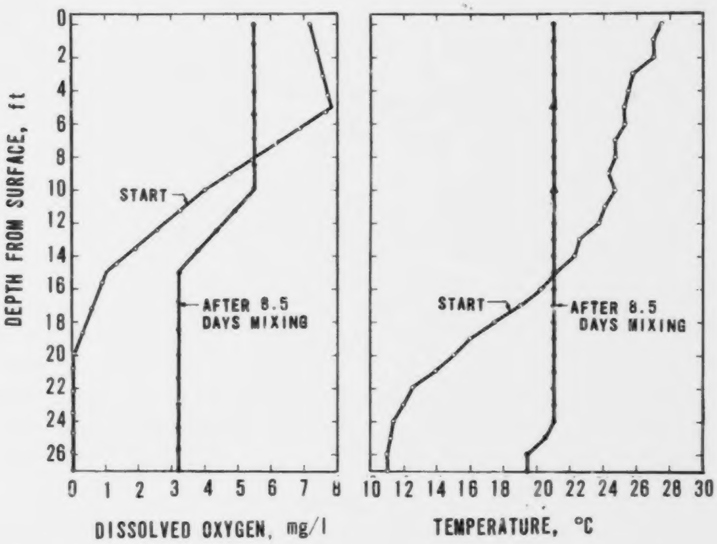

Figure 4. Artificial destratification of Vesuvius Lake, Ironton, Ohio. All data taken at pumping site.

mechanical equipment will be made on larger bodies of water.

Biologic Transformations of Nitrogen

Although the nitrogen concentration in most impoundments is relatively low, the changes from one nitrogen form to another caused by the biological population in the impoundment are important. Oxidation of ammonia consumes dissolved oxygen, reduction of oxidized nitrogen supplies oxygen, and nitrogen plays a vital role in the synthesis of new organisms, both bacteria and plankton. These transformations, studied by many investigators and summarized by Hutchinson,[10] are important to investigate if the future water quality in an impoundment is to be truly predictable.

One of the principal concerns in previous nitrogen transformation investigations has been the inability to account for all the nitrogen as it changes form, that is, the inability to make a nitrogen balance. This inability has been noted by Symons[18] in activated-sludge studies, by Robeck and co-workers in unpublished results taken from soil lysimeter studies, and by Leclerc[19] in studies made in streams. Theoretically, under completely aerobic conditions, analyzing for ammonia, nitrite, nitrate, and organic nitrogen concentrations should be sufficient to establish a nitrogen balance. Since the previous three studies were all made under generally aerobic conditions, the inability to make a nitrogen balance resulted either from micro-anaerobic environments in the system that caused some reduction of the oxidized nitrogen forms and associated loss in nitrogen gas, or from errors in analyses of nitrogen forms.

Investigations in this area have two purposes, 1) to show whether a nitrogen balance can be made under controlled, completely aerobic conditions and 2) to study the influence of environmental factors on the biologic transformations of nitrogen. The unique experimental and analytic techniques responsible for the success of this research are discussed.

The experimental method used is a modification of the one reported by Konecky, Kelly, Symons, and McCarty.[20] The apparatus consisted of several 5-gallon jars that had provisions for mechanical mixing through the top, introduction of air to maintain aerobic conditions, and a sampling siphon. The basic test medium was distilled water fortified with Difco Yeast Nitrogen Base, which supplied a wide variety of vitamins, minerals, and amino acids, as well as ammonia nitrogen in the form of ammonium sulfate. This was a good, clean, synthetic river water that was nutritionally balanced to permit rapid growth of autotrophic bacteria. In addition, the medium was free of substances which would interfere with any subsequent analyses.

The seed for these experiments was obtained from an operating bench-scale activated-sludge plant that was actively nitrifying. Mixed liquor was washed twice in distilled water to remove soluble carry-over products from the activated-sludge system and was blended to disperse the floc. Although the seed was not standardized, it was washed and was added in relatively low concentrations. This technique avoided the addition of unnecessary extraneous matter and approximated the volatile suspended-solids concentration that might be found in streams and impoundments.

Periodically during a test, samples were withdrawn and analyzed for pH, alkalinity, COD, dissolved oxygen, and all four nitrogen forms. In addition, at similar intervals, samples were withdrawn and the ammonia-oxidizing and nitrite-oxidizing microorganism populations determined. All of the nitrogen forms were determined either wholly or in part through the use of a Technicon AutoAnalyzer.* Details of this technique were discussed by Kamphake et al.[21] The ammonia- and nitrite-oxidizing microorganism populations were determined at various times during the experiments by the use of an MPN (5-tube replicate-serial dilution) procedure. This was an adaptation and extension of the method presented by Lewis and Pramer.[22]

An example of the type of results obtained during an individual test is shown in Figure 5. In this typical example, as well as in the other results, the classical nitrification pattern, oxidation of ammonia nitrogen and the production of nitrite and nitrate nitrogen, was clearly shown. In addition, the excellent recovery of nitrogen throughout the run was demonstrated by the ''ΣN'' curve, an arithmetic summation of the four forms of nitrogen on each of the days of analysis. Since this curve is nearly horizontal, all the nitrogen was continually accounted for throughout the experiment.

Since a nitrogen balance could be made, seven environmental influences were studied. The results of these experiments are summarized in Table 1.

Figure 6 shows an example of the correlation of the autotrophic microorganism population with the chemical

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*A product of Technicon Corporation, Tarrytown, N.Y.

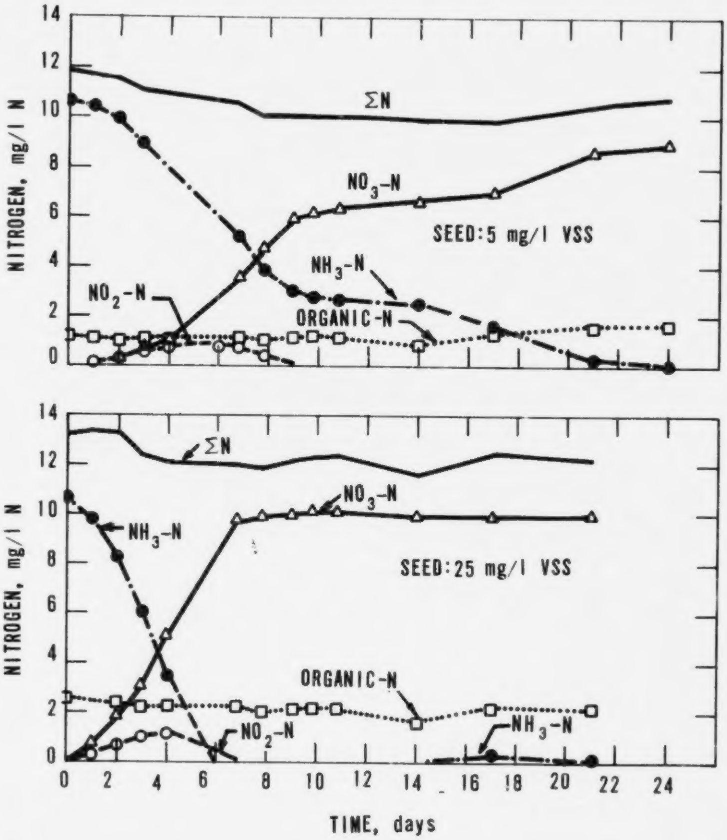

Figure 5. Influence of seed variation on nitrification, NH_3-N constant.

transformation of nitrogen in the same system. The ammonia-oxidizing population rises rapidly and maintains a comparatively high level during the active oxidation of ammonia, but, after the sixth day, when the ammonia concentration in the system has reached zero, the ammonia-oxidizing population starts to decrease. The nitrite-oxidizing population rises steadily and peaks on the seventh day, which corresponds to the completion of nitrification. After this time, the nitrite-oxidizing population begins to decline. Excellent correlation was made between the nitrifying populations and the chemical changes that occurred during nitrification.

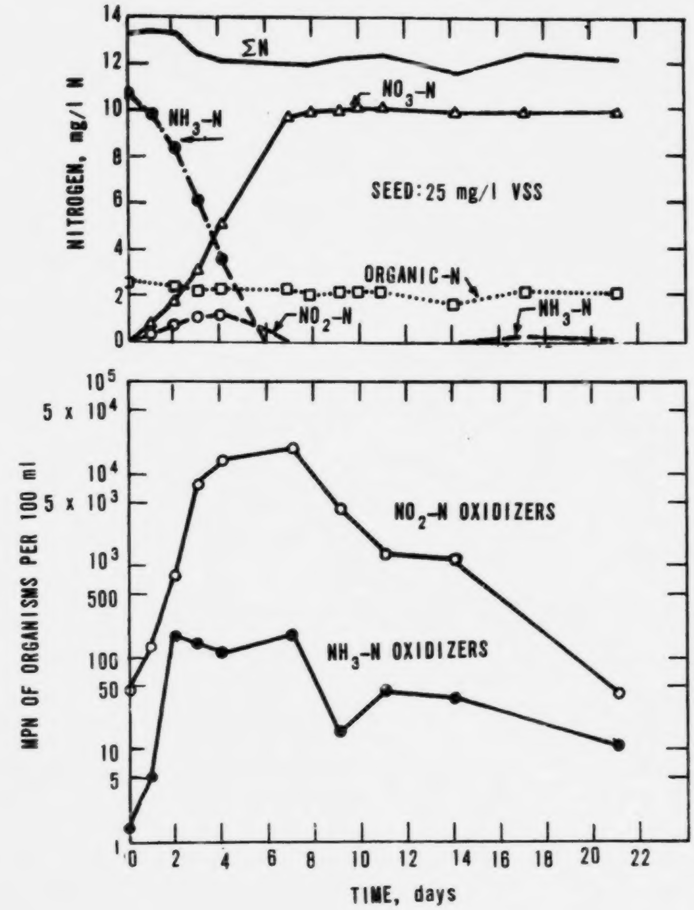

Figure 6. Correlation of bacterial and chemical data during nitrification.

Table 1. SUMMARY OF RESULTS ON BIOLOGIC TRANSFORMATIONS OF NITROGEN

Variable	Influence of variable on rate of NH_3-N oxidation
Initial seed concentration	Rate proportional to seed concentration
Initial NH_3-N concentration	No influence
Turbulence	Rate proportional to turbulence
Presence of NO_3-N	No influence
Presence of NO_2-N	Slows rate
Increase of concentration of vitamins and trace elements	No influence
Presence of glucose	No influence or slight increase

ACKNOWLEDGMENTS

The authors wish to acknowledge the work of the principal staff members of the Engineering Research Section of the Robert A. Taft Sanitary Engineering Center assigned to the project ''Effects of Streamflow Regulation on Water Quality.'' These are: Dr. William H. Irwin, Mr. Jack DeMarco, Mr. Jerzy Kurbiel, and Mr. Andre Cobet. Each contributed to the conception of the investigations described and to the taking of the data presented.

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PAPER NO. 16

ENHANCING BIODEGRADATION OF SYNTHETIC ORGANICS IN STRATIFIED IMPOUNDMENTS BY ARTIFICIAL MODIFICATION OF THE THERMAL PROFILE

J.M. Symons, J. DeMarco, W.H. Irwin, and G.G. Robeck

This paper was prepared to present some of the project research results to European investigators attending the Symposium on Lakes and Reservoirs, Lake Garda, Italy. By combining data from Paper No. 6 and Paper No. 8, this paper showed the potential benefit of reaerating the hypolimnion of reservoirs thereby permitting the rapid oxidation of organic materials. The oral discussion that followed presentation of the paper is also included.

The correct reference is:

Symons, J. M., J. DeMarco, W. H. Irwin, and G. G. Robeck. "Enhancing Biodegradation of Synthetic Organics in Stratified Impoundments by Artificial Modification of the Thermal Profile." Symposium of Garda - Hydrology of Lakes and Reservoirs, Publication No. 70 of De L'Association Internationale D'Hydrologie Scientifique (I.A.S.H.), 383-391, Gentbrugge, Belgium, 1966.

INTRODUCTION

The concept of using waters stored in impoundments as a source of dilution water for controlling downstream water quality developed in the United States about 1960.

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If a sufficient quantity of good quality stored water could be discharged into a river and, thus, increase the river flow at times of natural low flow, serious deterioration of water quality in the river could be prevented. This scheme was formalized in July of 1961 by Public Law 87-88, an amendment to the Water Quality Act of 1956.

This amendment directed all United States Federal construction agencies henceforth to consider the inclusion of storage capacity in their new impoundments for streamflow regulation for water quality control in cases where pollution could not be prevented by treatment. In addition, it directed an appropriate agency of the U.S. Government to conduct whatever basic and applied research was necessary to properly evaluate any water quality control project proposed by a construction agency.

Under this mandate, the Public Health Service organized a research project, ''Influence of Impoundments and Controlled Releases on Water Quality,'' in September 1962. The project staff's first choice for research concerned the changes that occur in water quality when water is impounded. Since impoundments will be the source of water in any streamflow regulation scheme, they must, at all times, contain good quality water for dilution. Research must be carried out, therefore, that will permit: 1) prediction of impounded-water quality in the future, 2) improvement of dam design for maintenance of good quality impounded water, and 3) development of methods to control water quality in impoundments that already exist.

This paper is a report on some of the research done on items 1 and 3 above, and is one of a continuing series of reports by the project staff on various phases of the overall project.[1,2,3,4,5]

LABORATORY STUDIES

The first phase of the research was carried out to determine the biodegradation patterns of certain synthetic organics in thermally stratified impoundments. Studies were conducted on a synthetic detergent, linear alkylate sulfonate (LAS), and a pesticide (herbicide) 2,4-dichlorophenoxyacetic acid (2,4-D) as the sodium salt. These two compounds were selected because they are both commercially important in the United States. LAS is the new detergent base that has replaced ABS in commercial production of U.S. household synthetic detergents; 2,4-D is one of the most common herbicides used in the United

States, and has been considered for use in impoundments to control water weeds.

The literature on the biodegradation of these two compounds is fairly extensive, so examples only are given. Klein and McGauhey[6] reported on the biodegradation of LAS in septic tank systems. They found that under aerobic conditions LAS was well, but not completely, metabolized, while under septic conditions metabolism was greatly reduced. Faust and Aly[7] reported the persistence of 2,4-D for 120 days in aerated river water, while Steenson and Walker[8] reported on the persistence of 2,4-D in anaerobic soil for 31 days. Since no literature could be found that related to the relative behavior of synthetic organics in the two environments existing in stratified impoundments, studies were undertaken to determine this behavior, at least qualitatively.

The studies were performed in a laboratory apparatus designed to simulate the thermal and dissolved oxygen profiles found in a typical stratified reservoir. The apparatus (Figure 1) consists of an aluminum tank, 3 feet (0.91 meters) in diameter and approximately 11 feet (3.36 meters) tall, which holds 500 gallons (2,000 liters) of test media. The upper two-thirds of the water column is maintained at approximately 23°C by a thermostatically controlled 12-kilowatt immersion heater, while the bottom third of the water column is maintained at about 12°C by a thermostatically-controlled refrigerator. Upper water is circulated by the continuous operation of a small mixer. Samples may be withdrawn from the upper and lower water through sample ports, and an injection port is available for the introduction of materials into the bottom-water zone.

In operation, the tank is filled with river water and allowed to stand for 24 hours to permit sedimentation of the larger suspended solids, which are then flushed out the bottom drain. The appropriate quantity of the synthetic organic under study is then added to the water and dispersed throughout with a high-speed mixer. The water is then thermally stratified, and an appropriate quantity of glucose is added to the bottom zone through the injection port and distributed throughout this zone by the small mixer near the bottom of the tank. The glucose aids the onset of low dissolved oxygen concentration conditions which are typical of deeper waters in natural stratified impoundments. Periodic sampling of the warm upper water where dissolved oxygen is present and the cold bottom water where dissolved oxygen is absent, or nearly

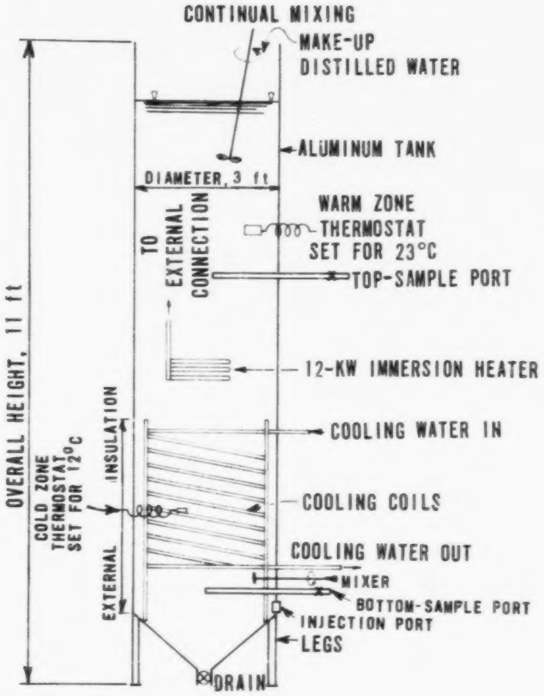
CONTINUAL MIXING
MAKE-UP DISTILLED WATER
ALUMINUM TANK
DIAMETER, 3 ft
TO EXTERNAL CONNECTION
WARM ZONE THERMOSTAT SET FOR 23°C
TOP-SAMPLE PORT
12-KW IMMERSION HEATER
OVERALL HEIGHT, 11 ft
COLD ZONE THERMOSTAT SET FOR 12°C
EXTERNAL INSULATION
COOLING WATER IN
COOLING COILS
COOLING WATER OUT
MIXER
BOTTOM-SAMPLE PORT
INJECTION PORT
LEGS
DRAIN

Figure 1. Simulated stratified impoundment.

absent, reveals the rates of disappearance of the synthetic organic under study. At the end of the test, a material balance is made to determine whether or not any disappearance measured in the test was caused by biodegradation or adsorption.

The data (Figure 2) clearly show that the environment in the lower waters of a stratified impoundment, where the temperature and the dissolved oxygen concentration are low, is adverse to rapid biodegradation of the two synthetic organics studied. In a field situation, these persistent materials might be unexpectedly discharged with water from low-level gates in a dam and, thereby, cause downstream pollution.

If, however, the thermal profile of a stratified impoundment could be altered in such a way as to maintain a warm temperature and a higher dissolved oxygen concentration throughout the impoundment depth, biodegradation of

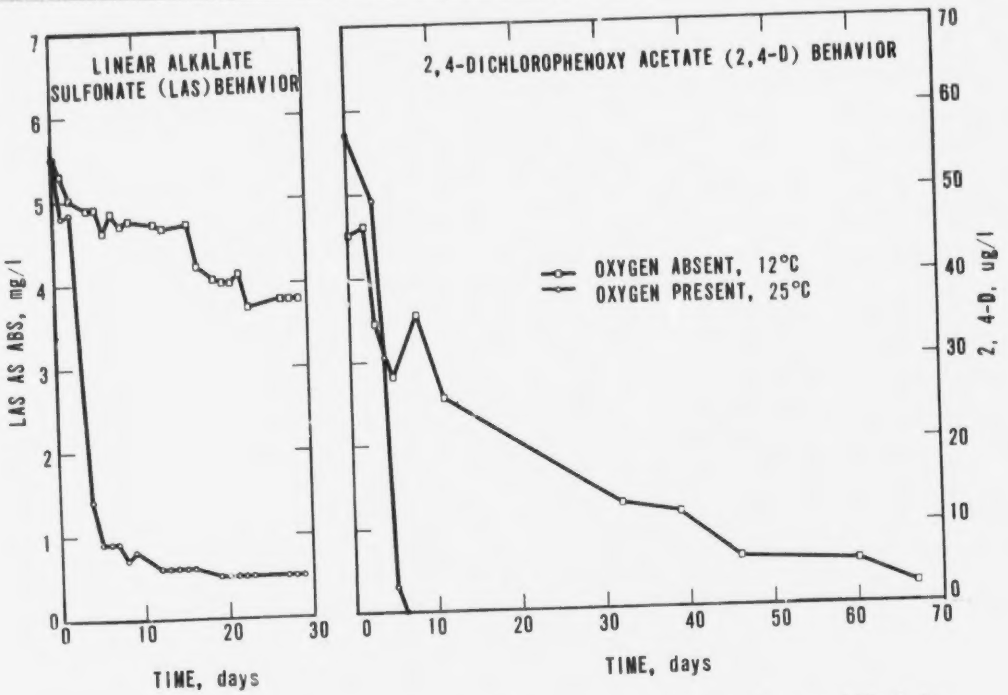

Figure 2. Behavior of synthetic organics in the simulated stratified impoundment.

these materials, and possibly others, would be enhanced, and the likelihood of downstream pollution would be minimized. This concept led to phase two of the study.

Three other synthetic organics (sodium lauryl sulfate, endrin, and lindane) were studied in this apparatus. Sodium lauryl sulfate was readily biodegraded in both the warm and the cold environments, while endrin and lindane were not materially biodegraded in either. These results are not reported in detail, since modification of the thermal profile of a stratified impoundment would probably not affect their biodegradation pattern.

FIELD STUDIES

Although the summertime density difference caused by the change in impounded water temperature from the surface to the bottom is sufficient to prevent the wind from completely mixing most impoundments, the actual difference in the weight of a unit volume of water at the extreme temperatures is quite small, generally less than 0.5 percent. Therefore, although an impoundment might contain a large volume of water, the actual work which must be applied per unit volume to raise the water from the bottom to the surface is quite small. The application of sufficient energy to a stratified impoundment to cause it to become completely mixed is therefore a possibility.

In light of the results of the previous section, complete mixing would have a very favorable influence on the rate of biodegradation of some synthetic organic materials. In addition, other improvements in water quality would arise from such a destratification operation. These include the elimination of hydrogen sulfide and the precipitation of any soluble iron and manganese that might have dissolved under low oxygen conditions. This portion of the paper describes the results of one method of artificially modifying the thermal profile of two stratified impoundments.

Although the literature on various attempts to artificially destratify impoundments is ample, only four are mentioned here. The first attempt was by Hooper, Ball, and Tanner,[9] who successfully destratified a Michigan lake in 1952. Koberg and Ford[10] used a diffused air system to destratify the Escondido, California, water supply reservoir. Bryan[11] summarized the applications of the Aero-Hydraulics Gun popularly known as the ''Bubble-Gun''. Ridley[12] reported on a proposed attempt to alter the position of the thermocline in one of the Metropolitan Water Board's reservoirs in London, U.K. Although his study is not complete at this time (1966) the device he describes is similar to the one used in this study.

A pontoon-mounted, mixed-flow pump, a schematic cross section of which is shown in Figure 3, was used in the investigation. The 12-inch pump was driven by a 16-horsepower gasoline engine, whose output was 12-horsepower at the operating speed of the pump for the Vesuvius Lake operation, and a 26-horsepower gasoline engine whose output was 21 horsepower at the operating speed of the pump for the Boltz Lake study. The suction line extended down to within a foot of the bottom where water was drawn in through a strainer, pulled up through the pump, and discharged at the surface.

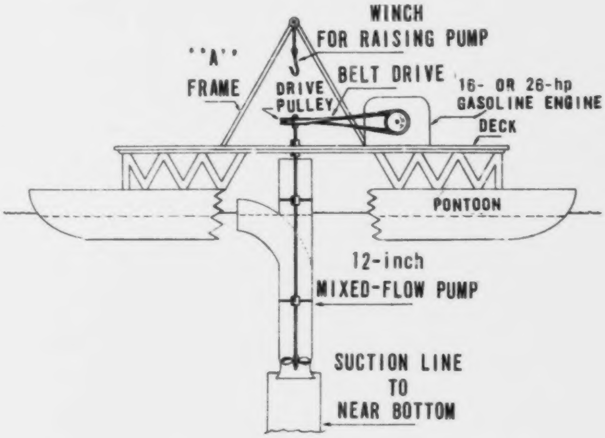

Figure 3. Design of pump, raft, and engine for destratification.

The ability of this device to function properly was demonstrated by destratifying three small (approximately 100 acre-feet) lakes in southern Ohio, in July 1964. After these three small-scale successes, destratification was attempted on two larger impoundments–Vesuvius Lake, Ohio and Boltz Lake, Kentucky. The outlines of these impoundments are shown in Figure 4. Pumping operations began in September 1964 in Vesuvius Lake and in August 1965 in Boltz Lake. The influence of the mixing operations on the temperature and dissolved oxygen concentration profiles in Vesuvius and Boltz Lakes is shown in Figure 5. These data show that the waters in the lower portion of the impoundments were both warmed and oxygen-

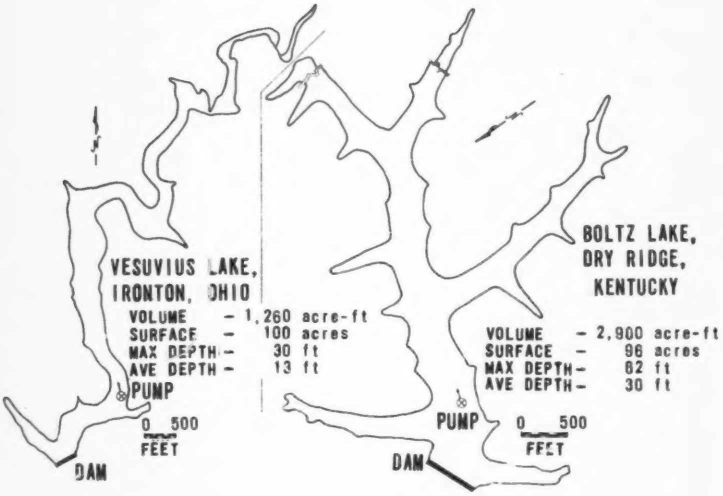
VESUVIUS LAKE,
IRONTON, OHIO
VOLUME - 1,260 acre-ft
SURFACE - 100 acres
MAX DEPTH - 30 ft
AVE DEPTH - 13 ft
PUMP
0 500
FEET
DAM
BOLTZ LAKE,
DRY RIDGE,
KENTUCKY
VOLUME - 2,900 acre-ft
SURFACE - 96 acres
MAX DEPTH - 62 ft
AVE DEPTH - 30 ft
PUMP
0 500
FEET
DAM

Figure 4. Outline of impoundments destratified.

ated by the pumping operation, which in Vesuvius Lake took 8-1/2 days and in Boltz Lake, 38 days. Oxygenation of the bottom waters in Boltz Lake was only slight. Apparently, the turnover time must be fairly short if the dissolved oxygen concentration in the lower zones of stratified impoundments is to be greatly increased.

Very little data, other than temperature and dissolved oxygen concentration, were collected for Vesuvius Lake to demonstrate the influence of mixing on water quality parameters, but several water quality parameters were measured in Boltz Lake throughout the pumping operation. Profiles of the water quality parameters that might be influenced by the reoxygenation of the lower waters of Boltz Lake are shown in Figure 6. The most significant changes occurred at the 30-foot level, where a rather marked increase in dissolved oxygen concentration was observed. Although the total COD concentration did not change much, sulfides were largely eliminated, and the manganese concentration was significantly reduced. Although no direct measurement of enhanced biodegradation of synthetic organics was obtained in the Boltz Lake experiment, in general, the water quality was improved because of the mixing operation. This improvement, undoubtedly, would have been even more dramatic had the mixing occurred in a shorter time.

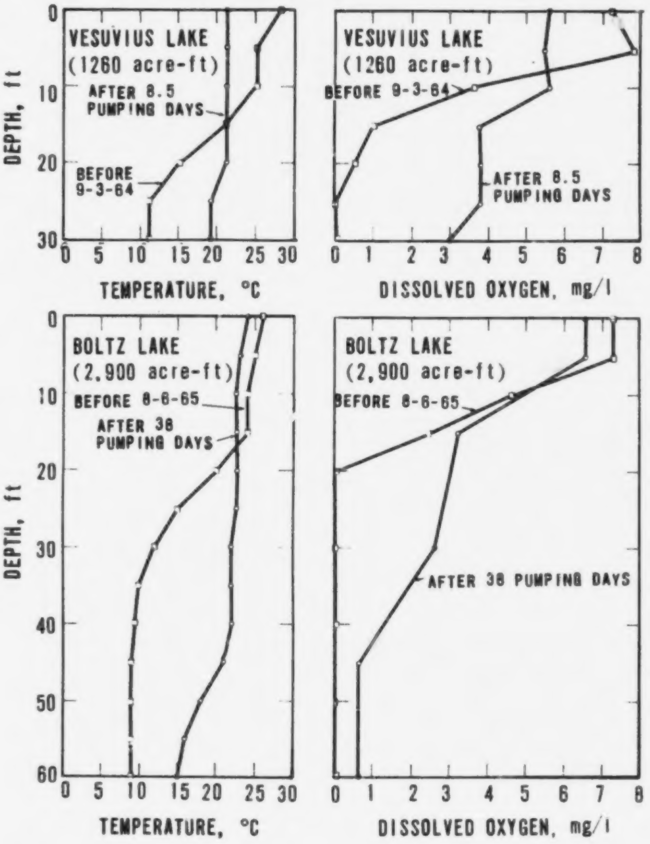

Figure 5. Results of mixing operations on Vesuvius and Boltz Lakes.

SUMMARY

A laboratory study, which measured the relative rates of biodegradation of linear alkylate sulfonate and 2,4-dichlorophenoxyacetic acid (sodium salt) in a simulated stratified impoundment, showed that biodegradation is faster in a warm environment with oxygen present than in a cold environment without oxygen. Since a cold, anaerobic environment is typical of the lower waters in natural stratified impoundment, field studies were con-

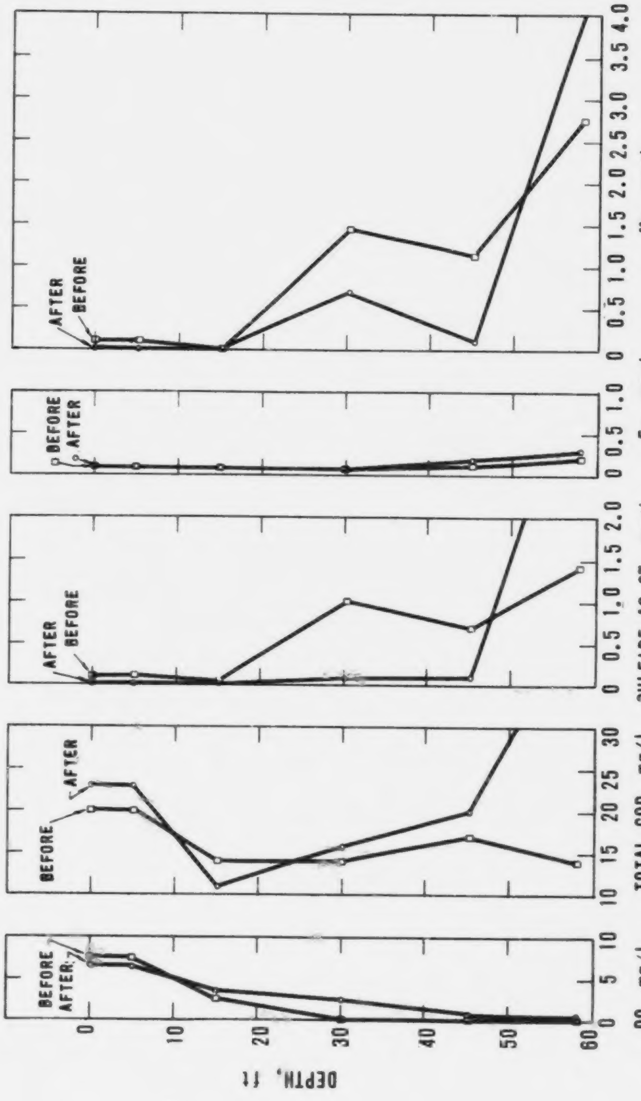

Figure 6. Influence of mixing on several water quality parameters in Boltz Lake.

ducted to determine the feasibility of artificially modifying the thermal profile of stratified impoundments to provide a more favorable environment for biodegradation of organic matter throughout the depth of an impoundment.

These studies have shown that a mechanical pump is an effective way to bring cold water from the bottom, discharge it at the surface, and thereby circulate the entire contents of a stratified impoundment. While mechanical pumping might not be the most efficient method, it improved the water quality throughout the depth of the impoundments studied by warming the water and providing dissolved oxygen at all depths. If this, or a similar device, were placed in existing storage reservoirs, the destratification it effected would help assure that good quality water is always available for release downstream.

ACKNOWLEDGMENTS

The authors wish to thank Engineering Activities staff members: Richard Shibiya for the LAS analyses; Donald Griffith, George Holtzer, and Cecil Pinkerton for the 2,4-D analyses; Lawrence Kamphake, George Holtzer, and Richard Shibiya for the analyses on Boltz Lake; and Glenn Gruber for help with the field work. In addition, acknowledgment is made of the cooperation of the personnel of the Ohio Department of Natural Resources, the National Forestry Service, and the Kentucky Department of Fish and Wildlife Resources.

QUESTIONS AND COMMENTS FROM THE AUDIENCE AND RESPONSES

Question by Mr. K. Szesztay, Civil Engineer, Section Head, Research Institute for Water Resources Development, Hungary:

Have you any information on evaporation reduction caused by decreasing the surface water temperature?

Answer – We have noted a decline in surface temperatures after mixing, and therefore we expect that the evaporation is reduced. We have not, however, calculated or attempted to measure the amount of this reduction.

Question by Mr. Herbert Moore, Consulting Engineer, Milwaukee, Wisconsin:

What form would the oxidized manganese, pesticides, and detergents take?

Answer – The manganese is oxidized from the +2 to +4 state. In the oxidized form, manganese is insoluble and would precipitate or adsorb. The pesticides, and synthetic detergents if biodegraded, would be oxidized to carbon dioxide and water plus some inorganic salts.

Question by Mr. P. Van der Burgh, Senior Engineer, Delta Service, Holland:

I wonder why the discharge pipe of the pump is so short? The effect will increase when lengthening the pipe and diverting the water in the upper layers in more directions. Experiments in the Netherlands show that it may be more effective to mix the water of different layers by putting compressed air into a perforated tube near the bottom of the reservoir. The flexibility of the system is of great importance.

Answer – The discharge pipe was short to promote turbulence in an attempt to mix warm top water with cold bottom water so that the pumped water would not sink back down to the bottom. We did, however, want to discharge the water in one general direction, along the long axis of the lake, in an attempt to promote beneficial inertial currents in the lake. These currents would then help mix the lake. We are familiar with attempts to destratify lakes with diffused air, and during the summer of 1966 have attempted this ourselves. We found this a very effective method of mixing, and shall be reporting on the details of that experiment in the near future.

Comment by Dr. V. M. Yevdjevich, Professor, Colorado State University, Fort Collins, Colorado:

The mixing of the cold and warm water of the surface by pure turbulence of the outflowing cold water from the pipe of the pump is very often ineffective. To increase the effectiveness of mixing, it is better to entrain the warm water before the pump (using the vacuum effect) and mix the two as the water flows through the pump.

Reply – We are not familiar with the effect that Dr. Yevdjevich mentioned, but we recognize the problem of decreasing the density of the pumped water to prevent its sinking back to the bottom of the lake. We agree that Dr. Yevdjevich's suggestion of pulling warm water into the suction side of the pump to promote better mixing is a valuable one.

Question by Mr. Blair Jones, U.S. Geological Survey, Washington, D.C.:

Three questions: a) In the graphs shown for profile distribution of manganese and sulfide, the bottom water concentrations increased, after treatment, to levels considerably above the natural, so as to appear to nearly overbalance the changes brought about above. Do you have any explanation for this? b) What is the distribution of the organic compounds of interest in the reservoir? Could the bulk of the input have been adsorption load concentrated on the bottom? c) At what depth is the outflow in the reservoir studied?

Answer – a) Because the suction of the pump was a foot or two from the bottom, a small quantity of poor quality water was left in the bottom of the lake after mixing. Although the concentrations of sulfide and manganese were very high in this volume of water, the total quantities were so small that the number of pounds of unoxidized manganese and sulfide that remained in the lake after mixing was only a fraction of 1 percent when compared to the total pounds of these materials that were in the lake before mixing began. b) We are not sure of the distribution of the organic compounds in the lake. Certainly a large quantity of organic material, both natural and adsorbed, is concentrated in the bottom muds. c) In Boltz Lake, the dam has a surface spillway, but during the pumping operation the weather was so dry that little, if any, water escaped over the spillway.

Question by Mr. Neal Minshall, Hydraulic Engineer, Agricultural Research Service, Madison, Wisconsin:

I wonder if Dr. Symons is familiar with the work done by the Wisconsin Conservation Department in introducing oxygen or air near the bottom of a 100-acre lake in southwest Wisconsin. It has been successful in eliminating the thermal gradient. The purpose of this study was to increase the oxygen content in the lower part of the lake and determine its effect on algal growth.

Answer – We have been in fairly regular correspondence with Mr. Thomas W. Wirth, Warm Water Group Leader, Southern Area Headquarters, Conservation Department, State of Wisconsin. He has described to us the proposed project and his method of attack using the Aero-Hydraulics Gun. At this time, however, we have not seen a detailed report on the results of this experimentation.

Question by Dr. Adam Synowiec, Laboratory of Lakes Physical Geography, Polish Academy of Sciences, Poland:

A basically similar experiment was done in a lake in northeastern Poland by Olszewski in a series of investigations that covered several years. The hypolimnion water was withdrawn through a pipe from the deepest point in the lake. The epilimnion was thus made thicker and thicker, finally occupying the entire depth of the lake. The thermal, chemical, and biological conditions, and their changes, were controlled. The trend of water quality changes was found to be unsteady. A secondary thermocline was sometimes reformed, at least during some periods of the experiment. Dissolved oxygen losses, or even the absence of DO, was observed, and chemical constituent concentrations varied. The latter two items probably occurred as a result of intensified decomposition processes that were fed by the greater organic production in the thicker epilimnion.

I would like to ask you whether the greater organic production in the reservoir waters, which now have better trophogenic conditions, could be the probable cause for the increasing concentrations of some chemical constituents observed in your experiments in the bottom layers of water? Also, were you controlling the biological situation in the reservoir during your experiment?

Answer – We think the oxygen-demanding materials in the bottom muds of our lake were the probable cause of the high concentrations of manganese and sulfides in the lower layers of water. Our algal counts show that there was very little increased primary production in the epilimnion of the lake after destratification. Also, we were not controlling the biological situation in our lake during the experiment.

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PAPER NO. 17
THE PROJECT REPORT

J.M. Symons

This paper, which is a combination of two papers, summarizes the entire project and is therefore, the project report. The first part of the paper, requested by the sponsors of the Annual Sanitary Engineering Conference at the University of Kansas, summarizes the controlled experiments on the behavior of synthetic organics; the nitrogen cycle in impoundments; and the measurement of the dissolved oxygen budget. The second part of this paper, requested by the sponsors of the Seminar on Reservoir Destratification, California Department of Water Resources, summarizes the field studies on artificial destratification as a water quality control technique. The discussion that follows this part of the paper was transcribed from the remarks that followed the oral presentation in Los Angeles, California. Twenty specific conclusions from all phases of the project are presented at the end of this paper.

The correct reference citations are:

Symons, J. M., W. H. Irwin, J. DeMarco, and G. G. Robeck. "Effects of Impoundments on Water Quality: A Research Summary." Transactions of the 17th Annual Sanitary Engineering Conference. The Bulletin of Engineering and Architecture, No. 57, The University of Kansas, Lawrence, Kansas. 1967. pp. 28-36.

Symons, J. M. "Reservoir Water Quality Control by Artificial Destratification - A Summary Report." Presented at the Seminar on Reservoir Destratification, sponsored by the California Department of Water Resources, Jan. 26, 1968, Los Angeles, California.

INTRODUCTION

Streamflow regulation for water quality control, a scheme in which good quality stored water is released at times of low streamflow as a means of diluting and thereby improving water quality (see Figure 1), has been under active consideration in this country over the past several years. Although reservoir releases have long been used to supplement mainstream water quantity, their use specifically for quality control has been limited to date.

To be successful, any streamflow regulation scheme is predicated on the availability of good quality discharges from the storage impoundment. If poor quality water is discharged into the main stream, deterioration of mainstream quality occurs and the water quality control scheme fails. Degradation of downstream quality because of poor quality impoundment discharges has been noted in various

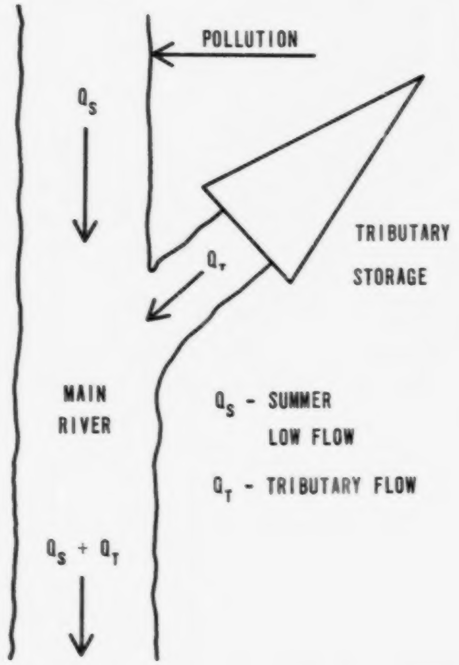

Figure 1. General scheme for streamflow regulation.

locations in the United States and has been documented in two locations by Ingols[1,2] and Fish.[3] Thus, research in this area, originally by the U.S. Public Health Service and currently (1967) by the Federal Water Pollution Control Administration staff in Cincinnati, Ohio, gave top priority to impoundment influences on water quality. This paper discusses in general terms the changes that may occur when a free-flowing stream is impounded and summarizes the research work carried out by the U.S. Public Health Service and the Federal Water Pollution Control Administration.

Effects on Water Quality

Churchill,[4] Love,[5] and Woodward and LeBosquet[6] have detailed the possible beneficial and detrimental effects that may be realized when a free-flowing stream is dammed. These effects are summarized in Table 1. Beneficial effects include: 1) reduced turbidity because of low velocities and long detention times, 2) reduced hardness caused by algal production of carbon dioxide and subsequent precipitation of calcium carbonate, 3) reduced color and BOD caused by long detention times, permitting biodegradation, 4) reduced coliform density caused by increased detention time, permitting natural die-off, and 5) equalizing action, smoothing high concentrations of various incoming pollutants.

Table 1. INFLUENCE OF IMPOUNDMENTS ON WATER QUALITY

Possible benefits	Possible detriments
Turbidity reduction	Less mixing
Hardness reduction	Lower reaeration
Organic oxidation	Backup of pollutants
BOD reduction	Algal blooms
Color reduction	Aesthetics
Coliform reduction	Tastes and odors
Equalizing action	No bottom scour
	Thermal stratification
	Low dissolved oxygen
	Iron and manganese dissolution
	Hydrogen sulfide production
	Carbon dioxide increase
	pH reduction
	Organic persistence

Several detrimental effects that may occur when water is impounded include: 1) less mixing because reduced velocities may cause a waste that formerly was distributed throughout a stream to hug the shore line, 2) lower atmospheric reaeration caused by lower velocities and increased depth, 3) a backup of pollutants caused by a changing water surface profile, 4) increased algal problems, appearing as aesthetically displeasing scums, or taste and odor in the water, both of which are caused by the increased detention time in reservoirs, 5) no bottom scour because of increased depth, and 6) thermal stratification.

Thermal stratification is the major difference between streams and impoundments and is one reason that the great backlog of information developed on stream behavior is not strictly applicable to reservoirs and lakes. Thermal stratification has been thoroughly reviewed by Kittrell;[7] thus, only its effect on water quality shall be discussed here. Warm, light water lying over cold water acts as a lid on the system. This lid prevents the bottom, dense water from circulating to the surface where it would be reaerated. The natural oxygen demand of this water will remove the dissolved oxygen (DO) from solution. When the DO concentration goes to zero in the dense water zone, several reactions may take place. Under the anaerobic conditions, iron and manganese are reduced and go into solution; sulfates are reduced and odorous hydrogen sulfide is formed; excess carbon dioxide is created and the pH of the water is lowered; and aerobic biodegradation of organic materials ceases. All of these reactions are potentially detrimental to water quality; thus, when the bottom water of a stratified impoundment is released through power turbines, poor quality water may be discharged downstream.

The USPHS and FWPCA Research Program

Since September 1962, the USPHS and subsequently the FWPCA have been conducting research at Cincinnati, Ohio, on the influence of impoundments on water quality. The results to date (1967) are detailed in references 8 through 17. The major findings of this research are summarized in this paper. In general, this research is categorized as 1) investigative research, designed to better understand impoundment behavior, and 2) engineering research, designed to control impoundment water quality.

INVESTIGATIVE RESEARCH

Synthetic Organic Study

Because the bottom waters of a stratified impoundment often are anaerobic, those synthetic organics that normally are considered biodegradable may tend to persist for extended periods of time under anaerobic conditions. Thus, the discharge of bottom water through power turbines might release an unexpectedly large concentration of a given synthetic organic material. To test this possibility an apparatus was designed to duplicate thermally and biologically a stratified impoundment. This apparatus is schematically represented in Figure 2. The top waters were heated and circulated and the bottom waters were kept cold and stagnant. Figure 3 demonstrates that this equipment was able to duplicate on a small scale the temperature and DO regime found in a typical stratified impoundment in the field.

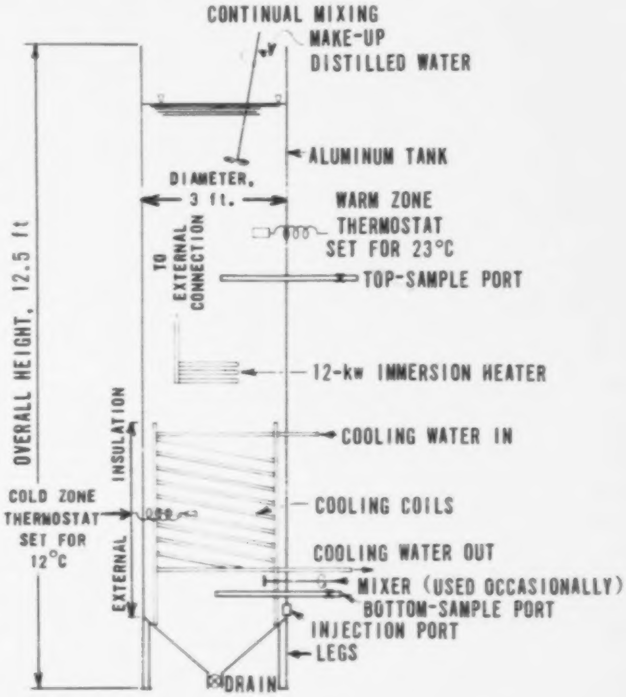
CONTINUAL MIXING
MAKE-UP DISTILLED WATER
ALUMINUM TANK
DIAMETER, 3 ft.
WARM ZONE THERMOSTAT SET FOR 23°C
TO EXTERNAL CONNECTION
TOP-SAMPLE PORT
12-kw IMMERSION HEATER
OVERALL HEIGHT, 12.5 ft
EXTERNAL INSULATION
COOLING WATER IN
COOLING COILS
COLD ZONE THERMOSTAT SET FOR 12°C
COOLING WATER OUT
MIXER (USED OCCASIONALLY)
BOTTOM-SAMPLE PORT
INJECTION PORT
LEGS
DRAIN

Figure 2. Simulated stratified impoundment.

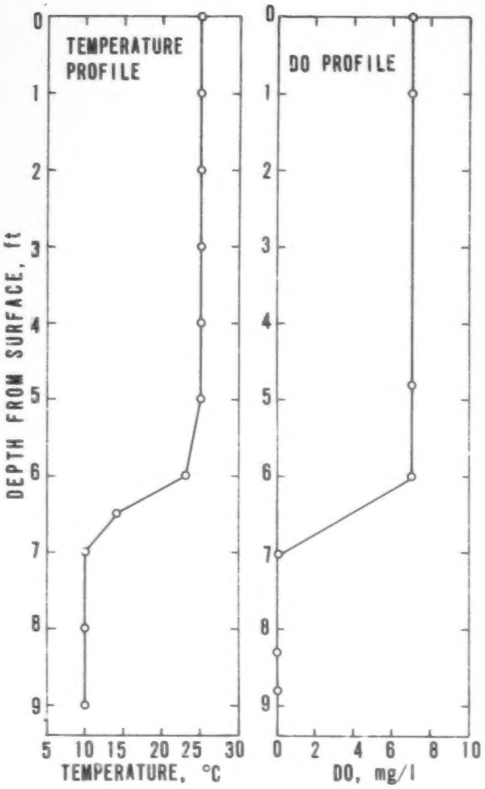

Figure 3. Typical temperature and DO profiles from simulated impoundment.

Three synthetic organics were tested: a very soft synthetic detergent, sodium lauryl sulfate (SLS); the new biodegradable base for detergents in the United States, linear alkylate sulfonate (LAS); and a commercially important pesticide, 2,4-dichlorophenoxy acetic acid (2,4-D). To conduct the test, the tanks were filled with water taken from the near-by Little Miami river. Each synthetic organic under study was added to the water and dispersed throughout with a high speed mixer. The water column was allowed to thermally stratify, and glucose was injected into the cold bottom water to hasten the onset of zero DO conditions. The disappearance of the synthetic organic under study was monitored by periodically withdrawing samples from the two temperature regimes.

The very soft SLS material was rapidly and completely biodegraded in both regimes in 1 day or less. Figure 4 shows that the lack of oxygen and the cold temperatures severely limited the rate of biodegradation for both the LAS and 2,4-D material. This confirmed the *a priori* suggestion that the bottom environment of a stratified impoundment was not conducive to biodegradation, and material, though aerobically quite biodegradable, might tend to persist under typical bottom water conditions. These experiments have been detailed in reference 15.

Nitrogen Studies

Because 4.57 milligrams of oxygen are required to oxidize 1 milligram of ammonia nitrogen to nitrate nitrogen, nitrification can create a severe drain on the DO resources in an impoundment. This study sought to determine the influence of various naturally occurring environmental factors on the nitrification reaction and, ultimately, how to predict nitrification.

To study biologic transformations of nitrogen properly, an apparatus and a set of analytical procedures had to be developed that would permit a continuous record of all of the nitrogen, as ammonia nitrogen was first changed to nitrite nitrogen and then to nitrate nitrogen. This was accomplished by studying nitrification in a clean, well-defined, inorganic medium, under completely aerobic conditions. This, combined with the use of the hydrazine method for nitrate analyses as detailed by Kamphake, Hannah, and Cohen[18] permitted the maintenance of a nitrogen balance throughout all of the various studies. A typical nitrification pattern, showing the excellent nitrogen balance, is seen in Figure 5. The volatile suspended solids (VSS) for these tests were obtained from a bench-scale activated sludge system that was actively oxidizing ammonia nitrogen.

Although a predictive method for nitrification was not found, some interesting results were obtained during this experimentation. For example, the direct relationship between increasing nitrifier population and increasing rate of nitrification was developed (Figure 6). Also, the initial rate of nitrification was shown to be independent of the original ammonia nitrogen concentration, at least through the range of 5 to 25 milligrams of NH_3-N per liter (Figure 7). These results mean that, similar to many biological reactions in the range studied, nitrification is a zero order reaction with respect to substrate level and a first order reaction with respect

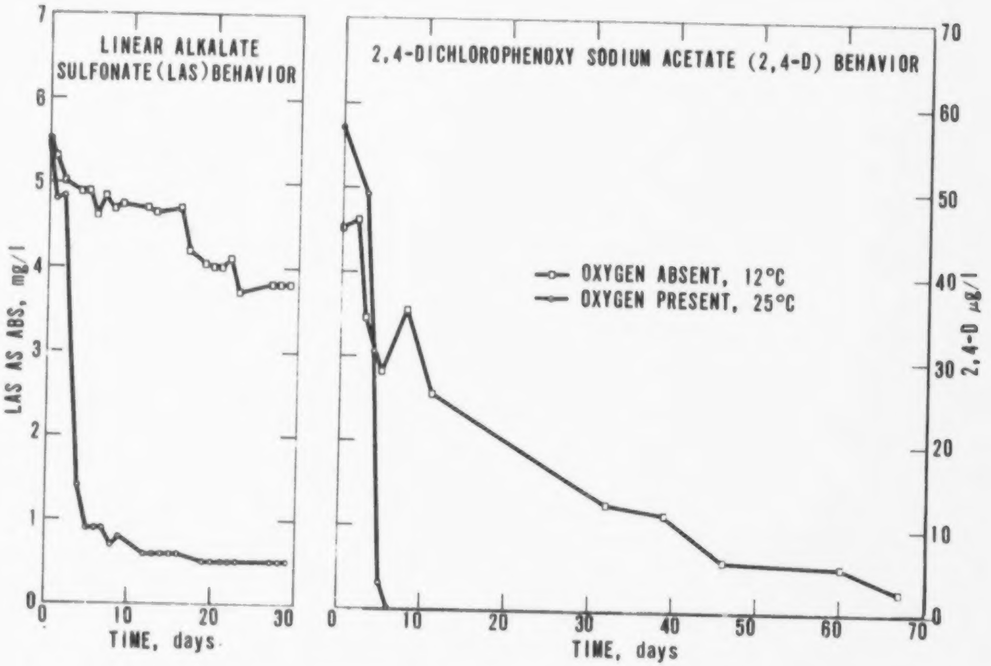

Figure 4. Behavior of synthetic organics in the simulated stratified impoundment.

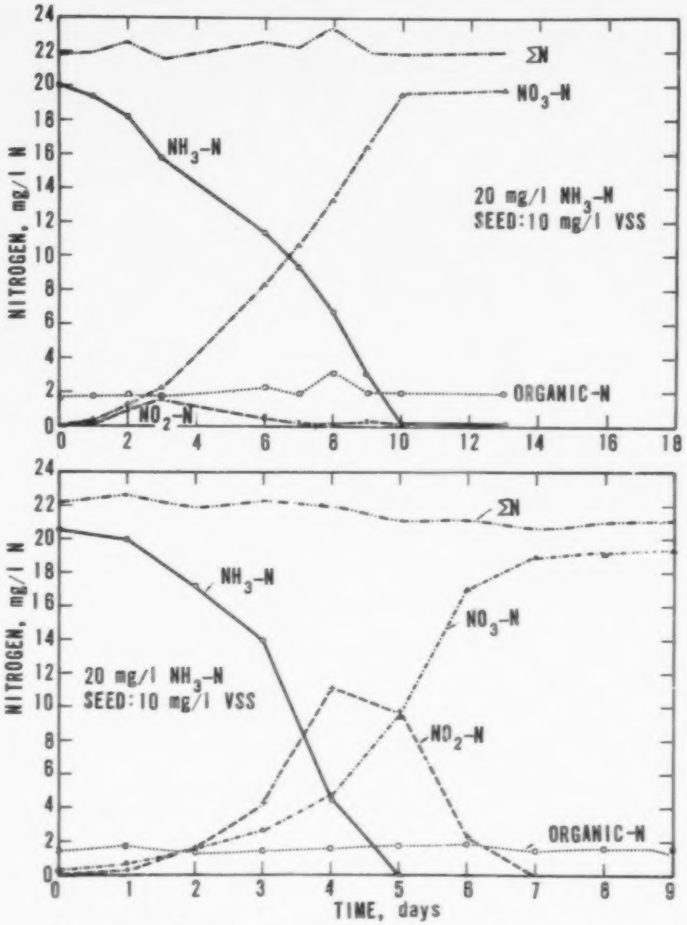

Figure 5. Typical nitrification patterns.

to microorganism population. These experiments have been reported in reference 16.

DO Budget

The relative role of algal photosynthetic production of oxygen and atmospheric reaeration in the DO budget of lakes and reservoirs is not well understood. Algae produce oxygen during the daylight hours but consume it at night, and any carbon dioxide that is fixed into organic form

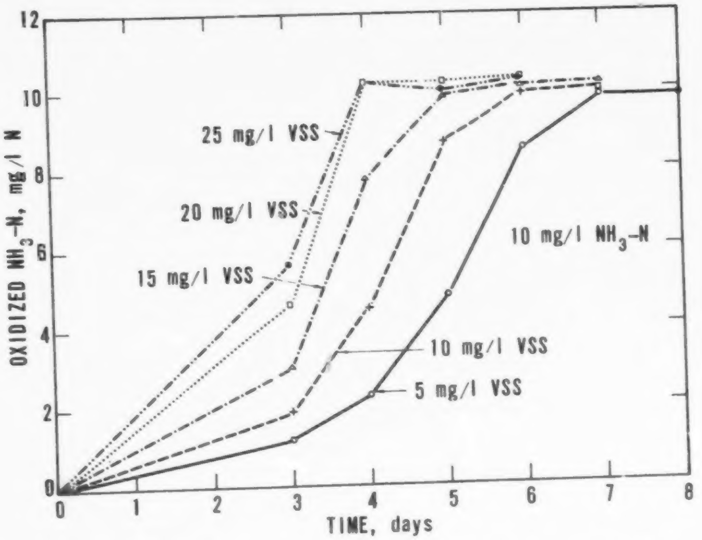

Figure 6. Influence of seed variation on nitrification.

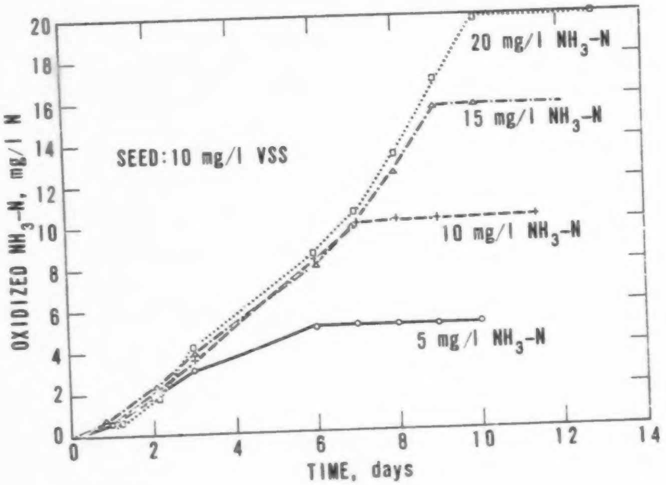

Figure 7. Influence of ammonia variation on nitrification.

with a corresponding production of oxygen will exert an oxygen demand later when the organic carbon is degraded back to carbon dioxide. Many studies have been made on the relative importance of stream velocity and depth to atmospheric reaeration, but most are inapplicable to an impoundment situation, because an impoundment usually has a much slower velocity and much greater depth.

Quite frequently the DO budget in impoundments is studied by the use of a technique called the light- and dark-bottle method. This technique has limitations, however. The five most commonly noted inaccuracies in the bottle technique are: 1) An unnaturally high surface-to-volume ratio in the bottles tends to increase the number of organisms over that existing in the natural environment. 2) The relatively long time the bottles must be left in place to permit the detection of changes in DO concentration may cause problems; if the external environment, wind or sunlight, changes significantly during this wait, the results obtained may not be applicable to the general case. 3) There is little or no mixing in the bottles, and this interferes with the flow of food to and the waste products away from the organisms; this may influence their metabolism and make the results somewhat atypical. 4) Bottles are usually filled on the boatdeck with water withdrawn from the impoundment; during this handling the algal population may be injured by exposure to sunlight. (This, of course, can be minimized by filling the bottles in the shade or at night, but, unless an impoundment is well-mixed, the bottled water, when returned to the test level for incubation, may not be truly representative of the water at the test level.) 5) Although the bottle technique does give some measure of the contribution of algal photosynthesis to the oxygen budget, it does little to measure the contribution of atmospheric reaeration to this budget.

The purpose of this study, therefore, was to design an apparatus that would minimize or eliminate the chief inaccuracies of the light- and dark-bottle method. Briefly, this apparatus makes use of three DO probes that can monitor the change of DO concentration with time in three different environments. One probe is placed in a black plastic container, one in a clear plastic container, and one in unconfined impoundment water.

Theoretically, the DO probe in the dark box should measure the decline in DO concentration caused by any oxygen-demanding materials in the water under study. The DO probe in the clear container measures the same oxygen demanding reaction, but this is offset by any oxygen production from photosynthesis. The DO probe in the unconfined water measures the oxygen demanding reaction, as well as any oxygen that is added to the water by photosynthesis and atmospheric reaeration. By measuring the change in DO concentration with time in these three environments, after the entire apparatus has been placed in a natural water under study, the magnitude of atmospheric reaeration and algal oxygen production may be obtained by properly comparing the various rates of change of DO with time.

The apparatus described was constructed (the clear plastic box in the open and closed position is shown in Figure 8) and electronic circuitry developed to amplify and record small changes in DO concentration so that such changes occurring over short periods of time would be detected. Although this apparatus was never studied in the field, it was tested in a large tank of artificial impoundment water. The results shown in Figure 9 demonstrate that this apparatus can monitor small changes in DO through an 8-hour period in the three different environments described above.

In an attempt to generalize, a mathematical model of the modified Streeter-Phelps type was chosen to represent the DO budget in an impoundment. The data from the three DO probes were then used to evaluate the constants in the mathematical model. The general mathematical model is given in Equation (1) below, and the solution for the results given in Figure 9 are presented in Equation (2).

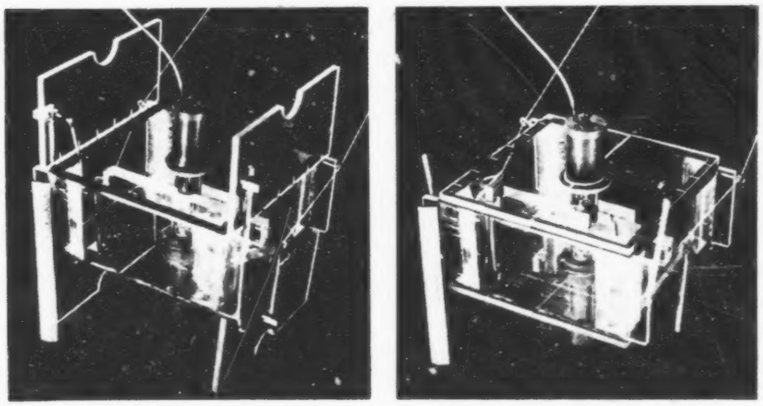

Figure 8. ''Light'' container and DO probe in open and closed position - note 12-inch scale in left foreground.

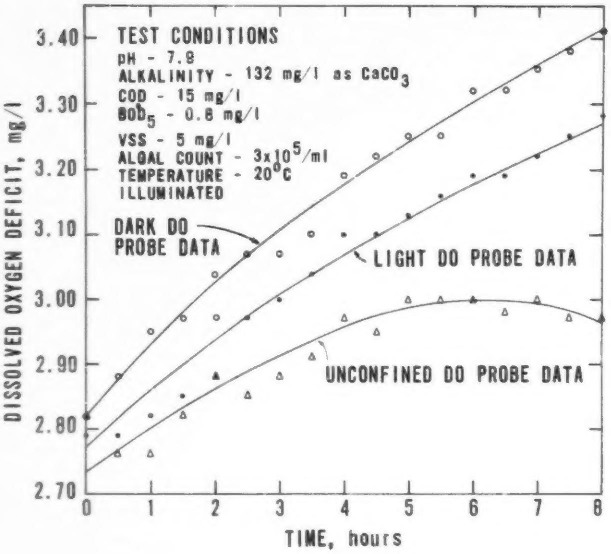

Figure 9. Data from special dissolved oxygen probe equipment.

$$D = \frac{k_1(L_\alpha)}{k_2 - k_1}\left[\exp(-k_1 t) - \exp(-k_2 t)\right] + \frac{(R-P)}{k_2}\left[1 - \exp(-k_2 t)\right] + D_\alpha\left[\exp(-k_2 t)\right] \quad (1)$$

Where:

D = DO deficit at any time
t = Time
k_1 = Deoxygenation rate constant
L_α = Total organic biochemical oxygen demand
k_2 = Atmospheric reaeration rate constant
R = Rate of oxygen demanded by the algal population
P = Rate of oxygen produced by the algal population
D_α = Initial DO deficit

$$D = \frac{(0.31)(0.51)}{0-0.15}\{\exp[(-0.31)(t)] - \exp[(-0.01)(t)]\} + \frac{-0.01}{0.01}\{1 - \exp[(-0.01)(t)]\} + 2.74\{\exp[(-0.01)(t)]\} \quad (2)$$

Figure 10 compares the data generated by Equation (2), and the actual data taken with the unconfined DO probe and shows their fairly close agreement. Complete details of the construction of the apparatus and the solution of the mathematical model are given in references 11 and 17.

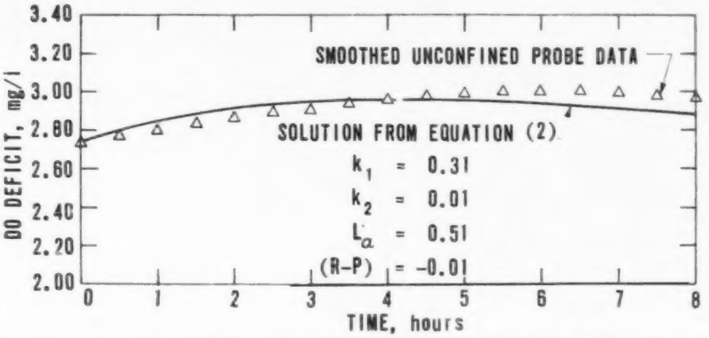

Figure 10. Comparison of Equation (2) with the unconfined probe data.

ENGINEERING RESEARCH

Authors Note: The paper presented at the California Department of Water Resources seminar begins here.

Many of the water quality problems that develop during the storage of water are related to thermal stratification. As mentioned above, this density-layering of water often leads to the disappearance of DO in the lower portions of the impoundment. The anaerobic condition created permits several undesirable reactions to occur, that, in turn, produce poor quality water. Studies were begun to determine if artificial destratification of lakes and reservoirs could prevent these reactions and thereby maintain good quality water vertically throughout the impoundment during a summer season. A theoretical analysis showed that the work required to lift a unit volume of cold water to the surface in a stratified lake was minimal (Figure 11), and that during such an operation, favorable density currents would develop, so as to drain all of the cold water initially present to the deepest portion of the reservoir (Figure 12). Based on these favorable concepts, artificial destratification was attempted, and the results are summarized below.

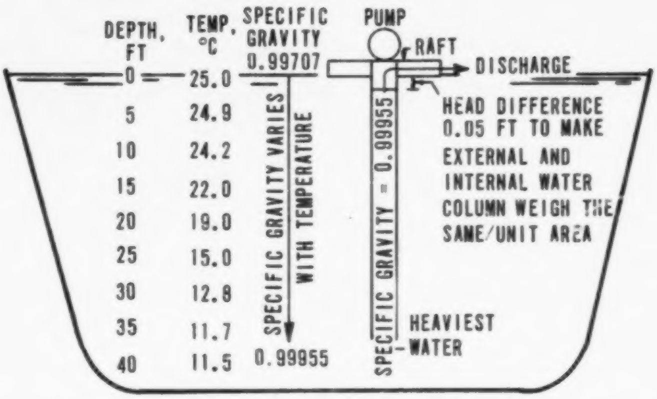
DEPTH, FT
TEMP, °C
SPECIFIC GRAVITY
0.99707
PUMP
RAFT
DISCHARGE
0
5
10
15
20
25
30
35
40
25.0
24.9
24.2
22.0
19.0
15.0
12.8
11.7
11.5
SPECIFIC GRAVITY VARIES WITH TEMPERATURE
0.99955
SPECIFIC GRAVITY = 0.99955
HEAD DIFFERENCE 0.05 FT TO MAKE EXTERNAL AND INTERNAL WATER COLUMN WEIGH THE SAME/UNIT AREA
HEAVIEST WATER

Figure 11. Theoretical elevation head needed to lift bottom water to surface-Lake Cowan, Ohio.

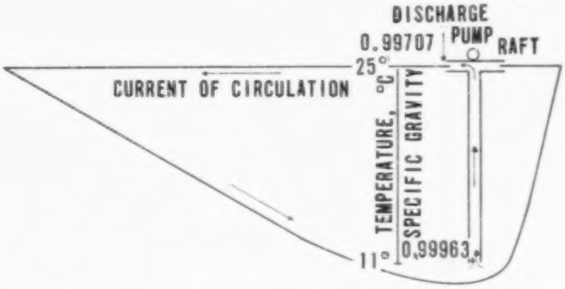
DISCHARGE
0.99707
PUMP
RAFT
25°
CURRENT OF CIRCULATION
TEMPERATURE, °C
SPECIFIC GRAVITY
11°
0.99963

Figure 12. General destratification scheme.

Artificial Destratification by Mechanical Pumping

The first destratification method investigated involved mechanical pumping of cold water from the bottom of a stratified water body and discharging it at the surface. The apparatus used for this investigation is shown schematically in Figure 13. A 12-inch mixed-flow pump driven by a gasoline engine was suspended over the deepest part of the lake from an anchored raft. The suction line extended to within 1 or 2 feet of the bottom and drew water from this depth for discharge at the surface.

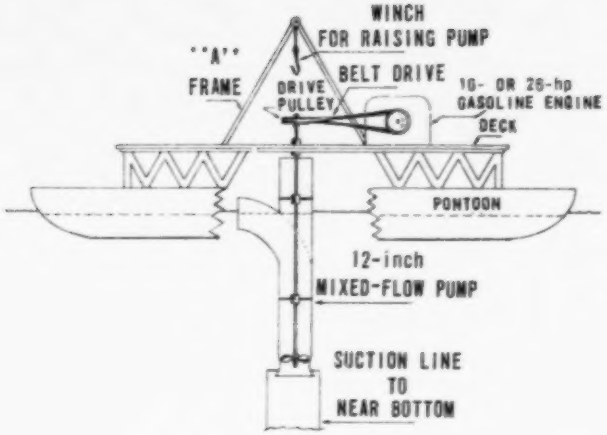
WINCH
FOR RAISING PUMP
"A"
FRAME
DRIVE
PULLEY
BELT DRIVE
16- OR 26-hp
GASOLINE ENGINE
DECK
PONTOON
12-inch
MIXED-FLOW PUMP
SUCTION LINE
TO
NEAR BOTTOM

Figure 13. Design of pump, raft, and engine for destratification.

1964 EXPERIENCE

In September 1964, this apparatus was placed in Vesuvius Lake, in south-central Ohio (Figure 14). Temperature and DO data (Figure 15) show that between September 4 and 17, the lake was made nearly isothermal and some DO appeared at all depths. The return toward original conditions 7 days after mixing stopped is also shown in this figure. Temperature profile data taken after 2.5 days of pump operating time and at the end of the test (8.5 days of pump operating time) at four locations in the lake (Figures 16 and 17) show that, except for a pocket of cold water trapped near the dam by a hump in the lake bottom, the entire lake could be influenced from one pumping location. Details of this study have been reported.[12]

1965 EXPERIENCE

In August 1965, this apparatus was placed in Boltz Lake, in northern Kentucky. The lake has a surface area of 96 acres and a volume of about 2,900 acre-feet. Bullock Pen Lake, a lake with relatively similar size and shape (142 surface acres and about 3,200 acre-foot volume), a few miles from Boltz Lake, was chosen as a "control" on the experiment (Figure 18). Important water quality parameters were measured weekly at various depths at the

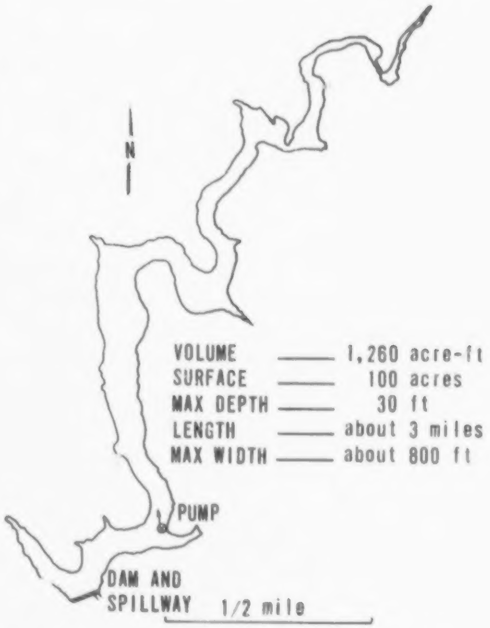
N
VOLUME ——— 1,260 acre-ft
SURFACE ——— 100 acres
MAX DEPTH ——— 30 ft
LENGTH ——— about 3 miles
MAX WIDTH ——— about 800 ft
PUMP
DAM AND SPILLWAY
1/2 mile

Figure 14. Outline and size of Vesuvius Lake near Ironton, Ohio.

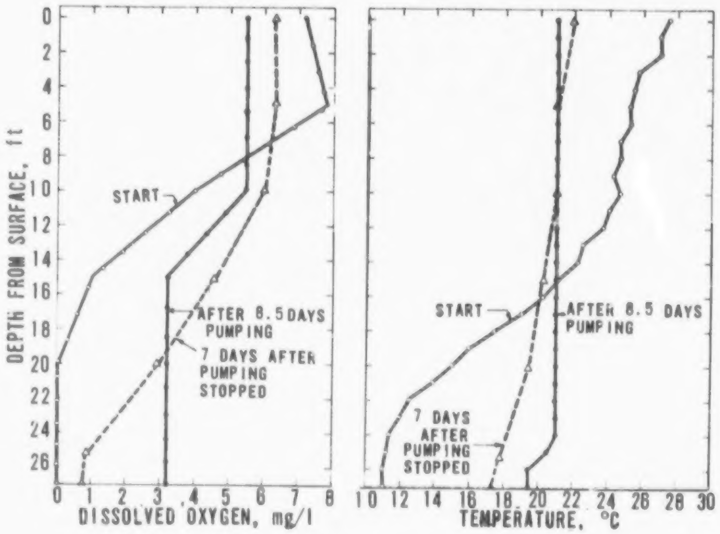

Figure 15. Artificial destratification of Vesuvius Lake. All data taken at at the site of pumping.

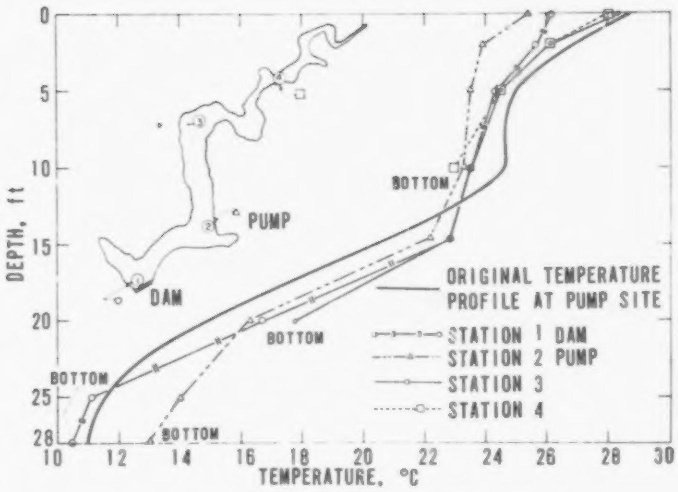

Figure 16. Change in temperature profiles at various stations after 2.5 days of pumping Vesuvius Lake.

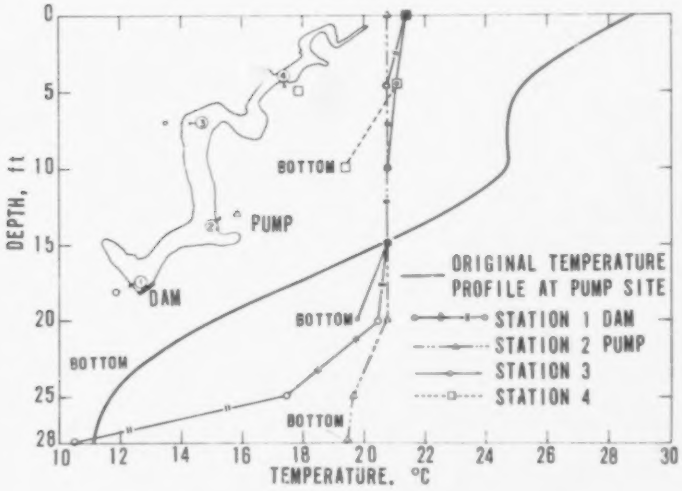

Figure 17. Change in temperature profiles at various stations after 8.5 days of pumping Vesuvius Lake.

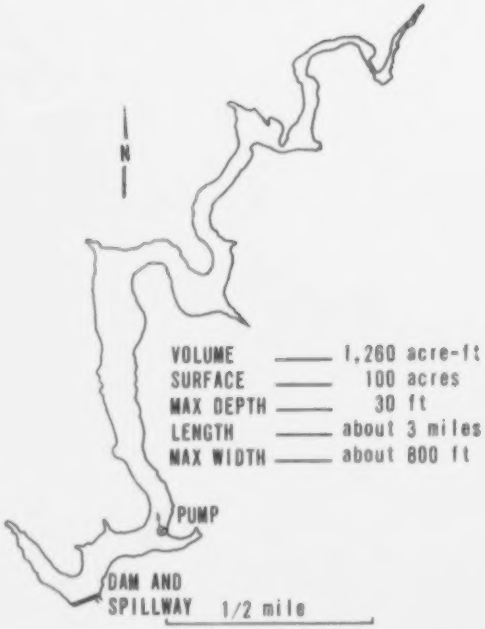
N
VOLUME — 1,260 acre-ft
SURFACE — 100 acres
MAX DEPTH — 30 ft
LENGTH — about 3 miles
MAX WIDTH — about 800 ft
PUMP
DAM AND SPILLWAY
1/2 mile

Figure 14. Outline and size of Vesuvius Lake near Ironton, Ohio.

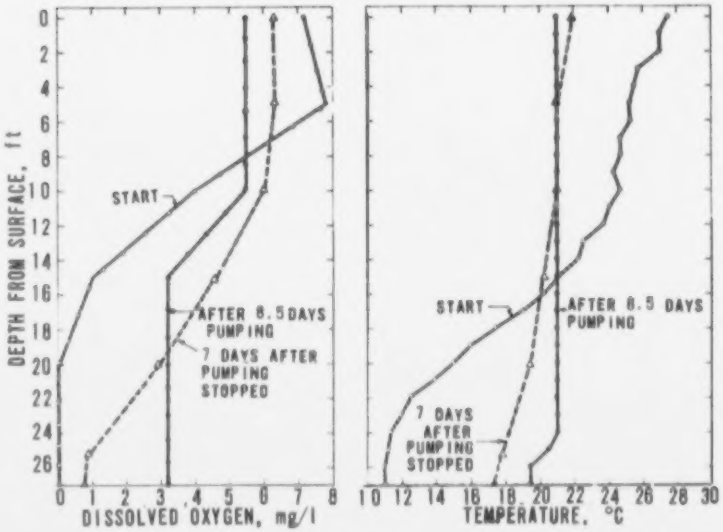

Figure 15. Artificial destratification of Vesuvius Lake. All data taken at at the site of pumping.

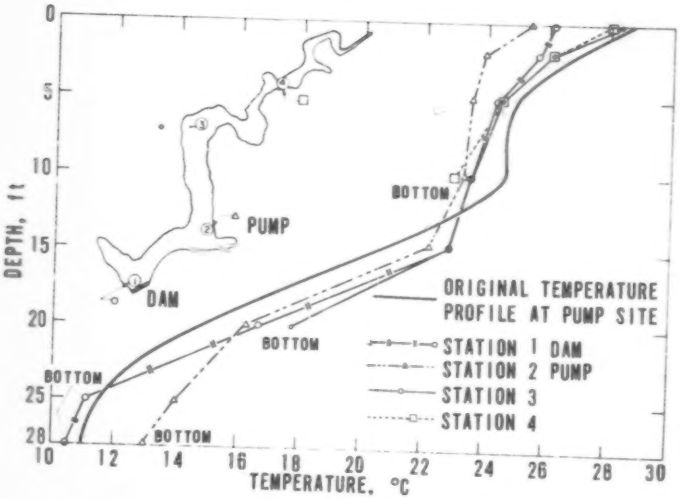

Figure 16. Change in temperature profiles at various stations after 2.5 days of pumping Vesuvius Lake.

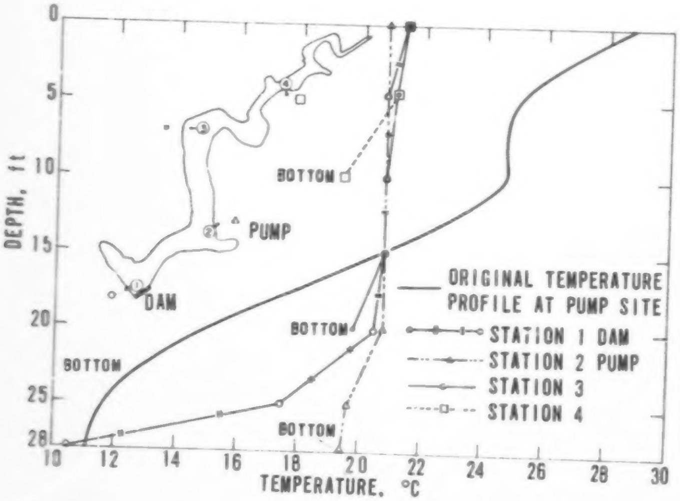

Figure 17. Change in temperature profiles at various stations after 8.5 days of pumping Vesuvius Lake.

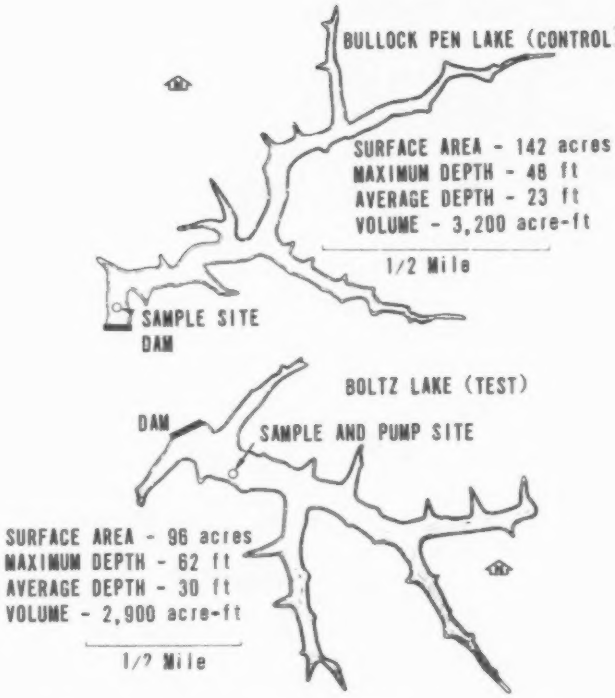
BULLOCK PEN LAKE (CONTROL)
SURFACE AREA - 142 acres
MAXIMUM DEPTH - 48 ft
AVERAGE DEPTH - 23 ft
VOLUME - 3,200 acre-ft
1/2 Mile
SAMPLE SITE
DAM
BOLTZ LAKE (TEST)
DAM
SAMPLE AND PUMP SITE
SURFACE AREA - 96 acres
MAXIMUM DEPTH - 62 ft
AVERAGE DEPTH - 30 ft
VOLUME - 2,900 acre-ft
1/2 Mile

Figure 18. Outlines of control and test lakes.

pump site in the test lake and near the dam in the control lake throughout the summer of 1965. A comparison of the progression of these parameters in the test and control lakes before the mixing of the test lake began in August 1965 showed that the two lakes were behaving in similar fashion.

The test lake was artificially destratified by pumping bottom water to the surface for 5 weeks during August and early September 1965. This artificial destratification improved the water quality in the test lake (Figures 19, 20, 21, and 22). Figure 19 shows that the lower water was warmed in the test lake, Figure 20 shows that DO was present at all depths after mixing, Figure 21 shows that the DO added to the test lake oxidized the sulfides that were originally present, and Figure 22 shows that the DO added to the test lake caused a reduction in the manganese concentration.

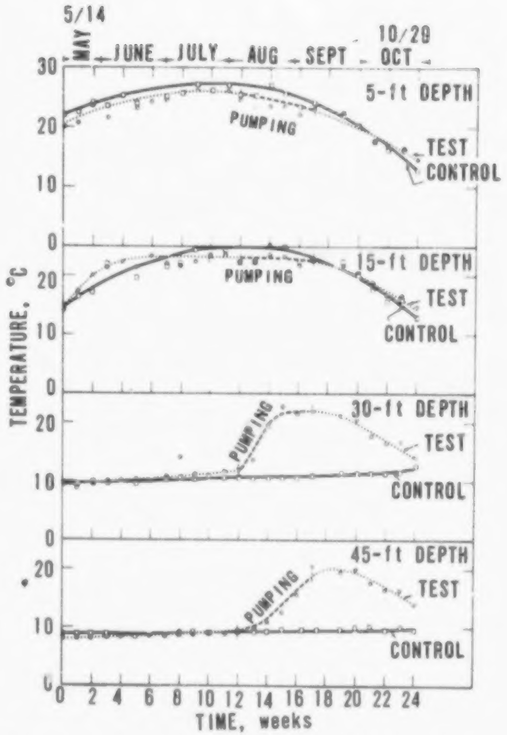

Figure 19. Comparison of changes in temperature with time at four depths in test and control lakes.

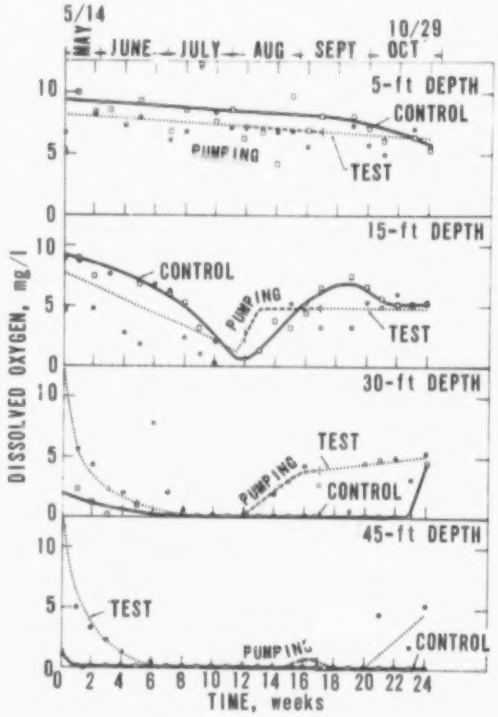

Figure 20. Comparison of changes in DO concentration with time at four depths in test and control lakes.

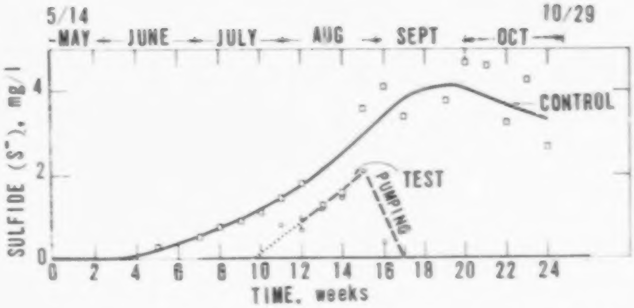

Figure 21. Comparison of changes in sulfide concentration with time at 45-ft depth in test and control lakes.

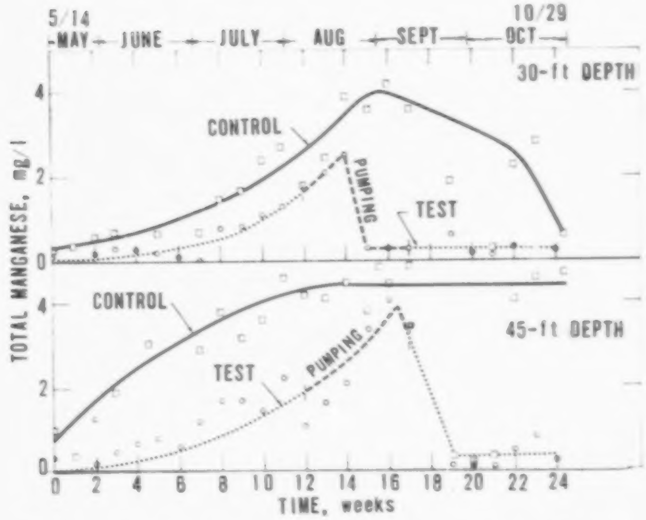

Figure 22. Comparison of changes in total manganese concentration with time at two depths in test and control lakes.

In addition, nutrient levels and algal populations were checked in the test and control lakes and these data showed that mixing did not bring nutrients to the surface in large amounts in the test lake and that the algal populations were not increased. Details of these tests have been reported.[14,17]

Artificial Destratification by Diffused-Air Pumping

Although the pumping operation described was effective in destratifying the test lake and improving its water quality, the energy input per unit volume was somewhat higher than originally expected. In an effort to provide the same improvement in water quality with reduced energy input, the same test lake (Boltz Lake) was artificially destratified with the use of a diffused-air pump during the spring and summer of 1966.

1966 EXPERIENCE

The equipment used for this operation was a 22.4 kilowatt-hour portable air compressor that delivered about 115 cfm of air at about 30 to 40 psi. The air was released through 16 porous ceramic diffusers spaced at 3-foot intervals in a cross pattern. This gave a cylinder of rising air approximately 40 feet in diameter that increased to about 100 feet in diameter at the surface. Figure 23 shows a schematic diagram of this equipment. In addition to testing whether this new method would be more effective than use of the mechanical pump, the 1966 destratification was started in the spring rather than in midsummer and mixing was performed periodically throughout the summer (called multiple mixing). This was done in an attempt to maintain good quality throughout a summer season rather than to improve water quality in midseason.[19] Also, during 1966 Falmouth Lake, Test Lake 2, was included for study (Figure 24). Table 2 gives the

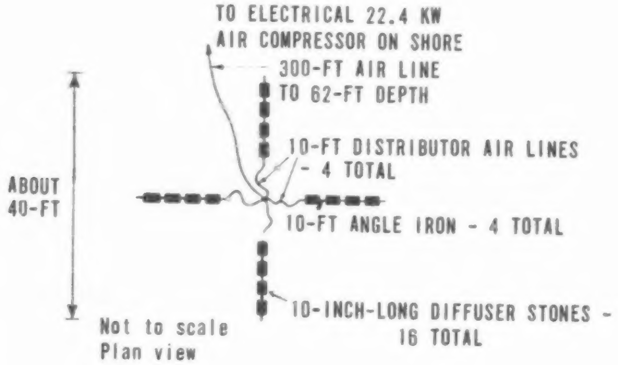
TO ELECTRICAL 22.4 KW
AIR COMPRESSOR ON SHORE
300-FT AIR LINE
TO 62-FT DEPTH
10-FT DISTRIBUTOR AIR LINES
- 4 TOTAL
ABOUT
40-FT
10-FT ANGLE IRON - 4 TOTAL
10-INCH-LONG DIFFUSER STONES -
16 TOTAL
Not to scale
Plan view

Figure 23. Schematic diagram of the diffused-air system.

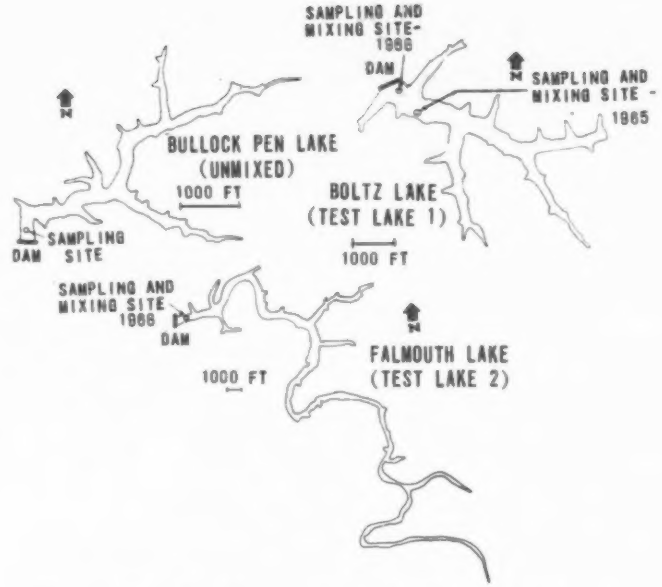
SAMPLING AND MIXING SITE-1966
DAM
SAMPLING AND MIXING SITE - 1965
N
BULLOCK PEN LAKE (UNMIXED)
1000 FT
BOLTZ LAKE (TEST LAKE 1)
1000 FT
SAMPLING SITE
DAM
SAMPLING AND MIXING SITE 1966
DAM
FALMOUTH LAKE (TEST LAKE 2)
1000 FT

Figure 24. Outlines of unmixed and test lakes.

Table 2. MORPHOLOGICAL CHARACTERISTICS OF STUDY LAKES AND DATES OF TESTING

Name	Designation	Volume, acre-ft	Max. depth, ft	Avg. depth, ft	Surface area, acres	Method of mixing	Mixing number	Mixing date
Bullock Pen Lake	Unmixed	3,200	48	23	142			
Boltz Lake	Test Lake 1	2,900	62	30	96	Mechanical pumping	1	Aug 6-Sept 10, 1965
						Diffused-air pumping	1	June 2-7, 1966
							2	June 21-24, 1966
							3	July 15-20, 1966
							4	Aug 18-23, 1966
Falmouth Lake	Test Lake 2	4,600	42	20	225	Diffused-air pumping	1	May 16-19, 20-23, 1966
							2	June 10-15, 1966
							3	July 1-8, 1966
							4	July 26-Aug 3, 1966
							5	Sept 8-13, 1966

mixing dates and summarizes the morphological characteristics.

Temperature

Figures 25A and B show the rising and falling pattern of the water temperatures in the unmixed lake at the 5-

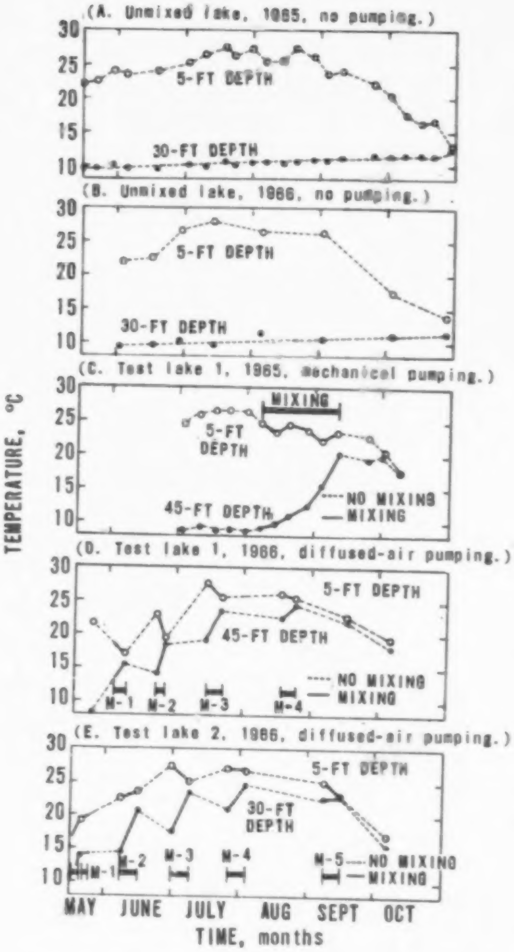

Figure 25. Comparison of temperature at the 5-ft depth and at 0.7 of maximum depth in the three study lakes.

foot depth during 1965 and 1966. Thermal stratification of lakes in the northern Kentucky area is shown by the low and relatively constant temperature that persisted throughout the summer season at the depth that is 0.7 of the lake's maximum depth. Temperatures at this depth were the same as temperatures at the maximum depth. For the unmixed lake this 0.7 depth is at 30 feet. Complete fall circulation is shown at the end of October in 1965 when the 5-foot and 30-foot temperatures became the same. In 1966, complete fall circulation occurred in mid-November; as a 2.5°C difference in the upper and lower temperatures existed at the end of October.

Figure 25C, included for comparison purposes, shows the increase in temperature at the 0.7 depth (same as bottom temperature) in the Test Lake 1 during mixing with the mechanical pump. At this time, there was a slight decline in surface water temperature. Thermal stratification was largely, but not completely, eliminated in the test lake during the 5-week mixing period.

Figures 25D and E contrast the temperature behavior at the 5-foot depth and at the 0.7 of maximum depth in the test lakes during 1966 when the diffused-air pumping with the multiple mixing technique was used. In every case, mixing caused a sharp temperature rise in the lower waters; in some cases, a temperature decline occurred in the surface water. The influence of solar radiation on the large surface area of Test Lake 2 is shown by the lack of any surface temperature decline during mixings 1, 2, and 4. Any decline in surface water temperature that occurs during artificial destratification retards evaporation.

Oxygen Resources

In an attempt to show the influence artificial destratification has on the DO resources in the two test lakes, a single oxygen parameter, the oxygen balance, was defined according to Equation (3).

$$[(\mathrm{DO}) + (-\mathrm{S}^{-}_{0_2}) + (-\mathrm{Fe}^{++}_{0_2}) + (-\mathrm{Mn}^{++}_{0_2})] = \text{Oxygen Balance} \qquad (3)$$

Where:

DO = Pounds of dissolved oxygen per million gallons below 30 feet

S_{0_2} = Pounds of the oxygen equivalent of sulfide (conc. X 2) per million gallons below 30 feet

$\mathrm{Fe}^{++}_{0_2}$ = Pounds of the oxygen equivalent of reduced iron (conc. X 0.29) per million gallons below 30 feet

$\mathrm{Mn}^{++}_{0_2}$ = Pounds of the oxygen equivalent of reduced manganese (conc. X 0.29) per million gallons below 30 feet.

When the oxygen balance is positive, DO usually is present and reduced materials are absent. A large negative oxygen balance usually means that large concentrations of some reduced materials are present and that DO is absent. Equation (3) was applied to the water below the 30-foot depth in all three study lakes because inspection of the data showed that in this geographic location thermal stratification caused a change in the DO resources in the water below this depth. Other oxygen ''sinks'' such as COD, BOD, NH_3-N, or the oxygen demand of the bottom muds were not included in Equation (3) because in these lakes analysis of the data showed that these factors did not deplete the DO resources very much. Consideration of these DO sinks might have to be made in other situations, however.

Figure 26 shows the general trend of the oxygen balance in an unmixed lake in this geographic location. DO exists

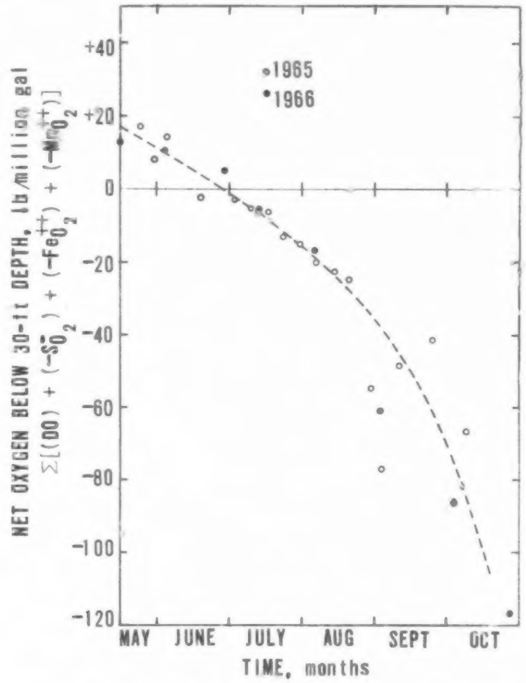

Figure 26. Oxygen surplus or deficit below 30-ft depth in unmixed lake.

in the water until early July when reduced materials tend to accumulate below the 30-foot depth. Figure 27 shows that in Test Lake 1, both in 1965 and 1966, artificial destratification improved the oxygen balance. The benefit of periodic mixing, as practiced in 1966, is shown by the almost complete absence of reduced materials below the 30-foot depth in Test Lake 1 during that year. Similar data for Test Lake 2 (Figure 28) show that the oxygen demanding reactions in this lake were much stronger than in Test Lake 1 and that reduced materials did build up during mid-summer in this lake, in spite of periodic mixings, each one of which improved the oxygen balance. More rapid, or more frequent mixing, or both, could have avoided this buildup of reduced materials.

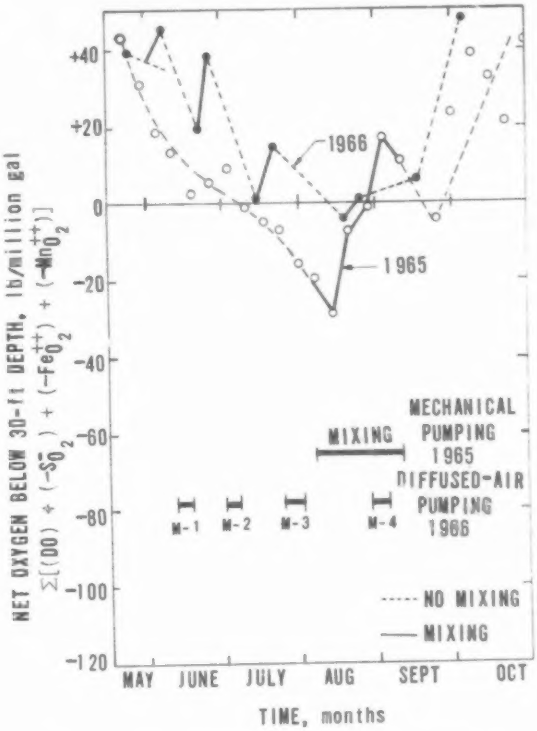

Figure 27. Oxygen surplus or deficit below 30-ft depth in test lake 1.

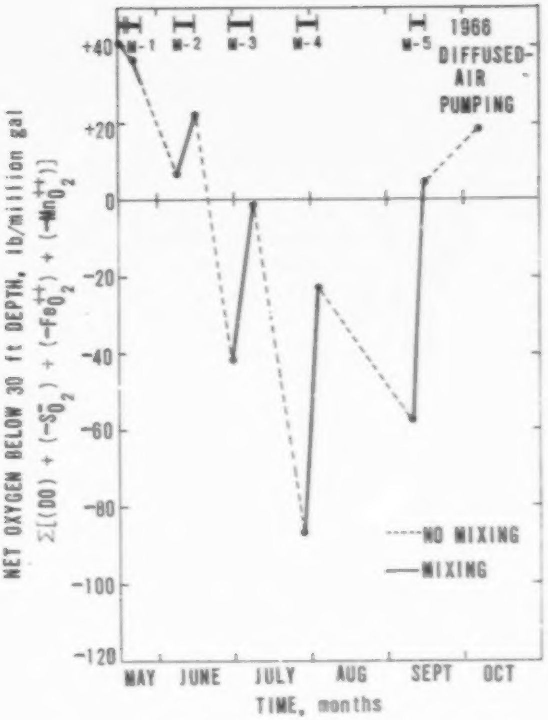

Figure 28. Oxygen surplus or deficit below 30-ft depth in test lake 2.

Plankton

Figure 29A shows that the surface plankton counts in the unmixed lake were similar in 1965 and 1966 and were between 1,000 and 3,000 organisms per milliliter from June to mid-September. Figure 29B shows a decline in plankton counts in Test Lake 1 during the 1965 mixing, and during diffused-air mixings 2, 3, and 4 in 1966. Figure 29C shows similar declines during 1966 in Test Lake 2 during mixings 3 and 5.

The reason for the declines in plankton counts is not known at this time (1968), but analysis of algal counts taken throughout the depth of the two test lakes showed that when declines in counts took place, they occurred

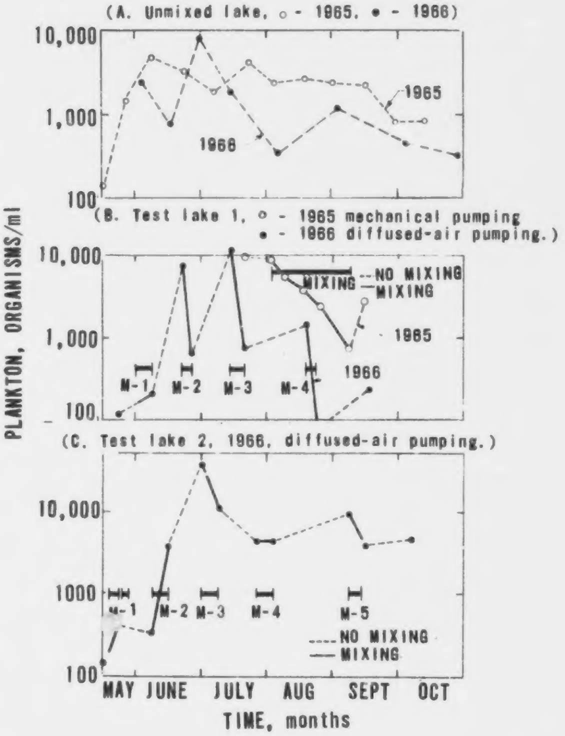

Figure 29. Plankton counts at the 5-ft depth in the three study lakes.

at all depths. The decrease in plankton population was, however, only temporary; Figures 29B and C show the increase in plankton counts after mixing stopped. These increases were not of "bloom" proportions, however, and excessive algal growth did not occur during any of the artificial destratification experiments.

Figures 30, 31, and 32 show the surface green and blue-green algal counts in the three test lakes. During the mixing periods in the test lakes, the blue-green algae counts declined more than the green algae counts (Figures 31 and 32). If this finding is not a coincidence and is proven reliable by further investigations, it indicates that a favorable predomination shift would occur during artificial destratification of a domestic water supply

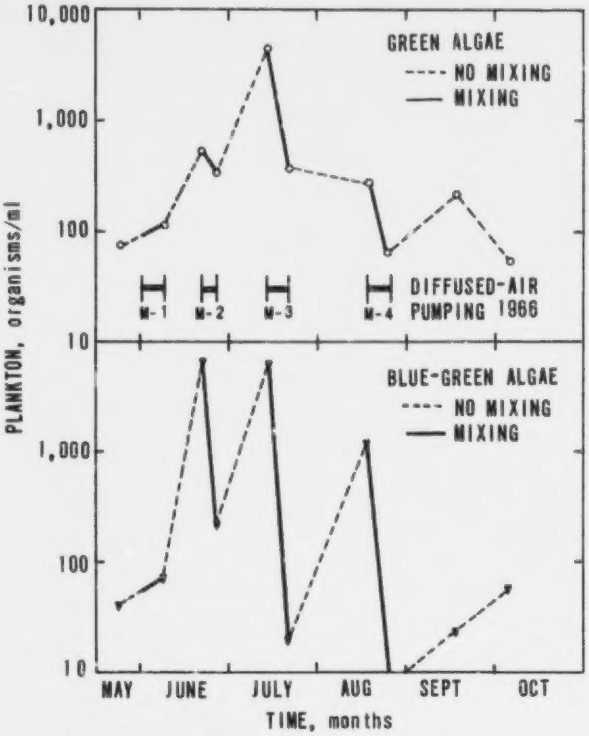

Figure 31. Green and blue-green plankton counts at the 5-ft depth in test lake 1.

Where:

t_1 = time of start of mixing

t_2 = time of end of mixing

Oxygen Balance = definition in Equation (3).

The units of OC are pounds of DO transferred per kilowatt hour.

$$\text{Destratification Efficiency (DE)} = \left[\frac{\text{Net Change of Stability from } t_1 \text{ to } t_2}{\text{Total Energy Input from } t_1 \text{ to } t_2}\right] \times 100 \tag{5}$$

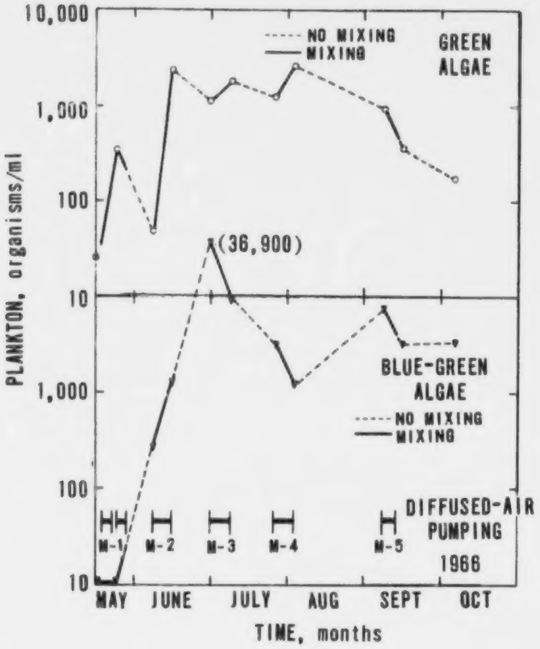

Figure 32. Green and blue-green plankton counts at the 5-ft depth in test lake 2.

Where:

t_1 = time of start of mixing

t_2 = time of end of mixing

Stability = minimum energy needed to mix the lake.

Two methods have been developed for evaluating Equations (4) and (5) (see Paper No. 12 in this Compilation). One method uses changes in oxygen balance and stability during a given mixing period divided by the corresponding energy input. The other uses the *area* between the control and test lake curves in the numerator for an entire season, while the denominator is the *area* under the cumulative energy input versus time curve over the same time period. This latter method has been named the ''area method.'' Suggestions for using these parameters, as calculated by the area method, for designing equipment are presented in the Appendix of Paper No. 14[20] of this Compilation. A performance summary for the mechanical pump and the diffused-air pump, from all experiments, calculated by both methods, is shown in Table 3.

Table 3. SUMMARY OF PERFORMANCE FACTORS DURING ARTIFICIAL DESTRATIFICATION

Lake	Dates of mixing	Method	Volume, acre-ft	Surface area, acres	Max. depth, ft	Oxygenation capacity,[a] lb DO/kw-hr	Destratification efficiency,[a] percent	Method of calculation
Vesuvius 1964	9/3-9/17	Mechanical pump	1,260	100	30	Unknown	0.4	Regular
						Unknown	Unknown	Area
Boltz 1965	8/6-9/10	Mechanical pump	2,900	98	62	1.3	0.2	Regular
						0.7	0.1	Area
Boltz (Test Lake 1) 1966	6/2-6/7	Diffused-air pump	2,900	98	62	1.6	1.5	Regular
	6/21-6/24					4.3	1.4	"
	7/15-7/20					3.3	1.5	"
	8/18-8/23						0.6	"
						1.7	0.5	Area
Falmouth (Test Lake 2) 1966	(5/13-5/19,	Diffused-air pump	4,600	725	42			
	5/20-5/23)					0.0	0.2	Regular
	6/10-6/15					1.4	0.9	"
	7/1-7/8					3.0	0.9	"
	7/26-8/3					4.2	0.8	"
	9/8-9/13					3.9	0.3	"
						0.02	0.2	Area

[a] THESE VALUES ARE INTENDED AS A ROUGH GUIDE ONLY AND ARE INFLUENCED BY MANY FACTORS.[19]

SUMMARY

Summarized, the benefits from artificial destratification were that 1) DO was found at all depths, 2) sulfide was oxidized, 3) iron was precipitated, 4) manganese was removed, 5) algae numbers were temporarily reduced, 6) blue-green algae were reduced in numbers more than green algae, and 7) oxidation of organic matter may have been enhanced. Before the mixing experiments began, increased algal growth was thought to be a possibility; the results, however, indicated substantial reducation in numbers during the mixing periods and regrowth that was not excessive. Also, organic oxidation was expected to to occur; the BOD and COD tests employed for this measurement were not sensitive enough, however, to make this determination, and oxidation, therefore, remained undetected. Whether fishery conditions were improved or reduced was not determined. Potential detriments to water quality from artificial destratification, which were considered *a priori*, were decreasing water clarity, which was not noticeable, and bringing nutrients to the surface, which occurred only in a few cases.

All indications are that an artificially destratified raw water reservoir would contain water that was easier to treat and therefore would produce an improved finished water for distribution to consumers. A survey conducted in the spring of 1969 showed that 11 water purveyors in various countries, 8 in the United States, have successfully used this technique for improving raw water quality, and at this time 10 others were considering the use of this method.

DISCUSSION FOLLOWING ORAL PRESENTATION AT THE CALIFORNIA DEPARTMENT OF WATER RESOURCES SEMINAR

QUESTION: From a practical standpoint, how large a reservoir does it now appear can be effectively destratified with a single unit?

ANSWER: It depends whether or not you can tolerate a zone at some distance away from your mixing device that is not mixed. Certainly, the bulk of the water is downstream near the dam where the reservoir is wide and deep, and if this is the area that we can actually affect from a single location, we may actually do 90 percent of what we want to do from a single location, even in a large reservoir. I have not yet seen the data taken by Federal Water Pollution Control Administration in Eufaula Reservoir in Oklahoma which gives the dimensions of their reservoir and the distance upstream which they did mix. I gather from qualitative remarks made at the ASCE Specialty Research Conference, January 21-24, 1968, Portland, Oregon, that the reservoir was some 30 miles long and that they were able to mix upstream 15 miles or so. I may be wrong on the details, but we do have to determine whether or not we can influence the bulk of the poor quality water. If so, I think we can operate from a single location.

I don't think that depth is any detriment to mixing, but when you are comparing pumping and diffused-air systems, you must remember that you have to compress the air to overcome the pressure at the bottom, whereas with a pump you can pump from any depth against approximately the same head because you are pumping against the same density difference. For very deep installations, a mechanical pump may turn out to be a more effective device than a diffused-air system to avoid this high compression of air. The diffused-air system appeals to many people because it can be installed very simply after a dam is finished–just a compressor on shore and some diffusers in the bottom. It may turn out that for large and deep installations some axial-flow, large-volume, low-head pump actually designed into the project may be the best way to handle this situation.

QUESTION: What was the rating of the air compressor in cfm and lb/in^2 and what was length of lake and maximum capacity in acre-feet used in the diffused-air pump experiment?

ANSWER: The lake mixed in 1965 and 1966 had a volume of about 2,900 acre-feet—a small lake, 62 feet in maximum depth, and approximately 1-1/2 miles in length. The air compressor delivered about 115 cfm at just enough pressure to overcome the hydrostatic head of 62 feet of water.

QUESTION: What will be the effect of destratification on use of lake surfaces for recreation in regard to pathogens and public health?

ANSWER: We do not know much about the behavior of bacterial and viral pathogens or bacteria and virus in mixed reservoirs. This is something that we have to look into. I, frankly, don't anticipate any problem in the water surface as a hazard to recreationists. We did have a question brought up at the Portland ASCE meeting about cooling the surface sufficiently to cause the water skiing and swimming to be less pleasant, and this is a possible disadvantage of destratification from a recreation point of view.

We are most interested in survival of *Salmonella* in the muds. Generally, muds are overlain by anaerobic water, and we wonder if their survival will be enhanced by being overlain with aerobic water; if this is true, we want to know about this as this would be another factor to consider. So, we are very interested in survival of enteric pathogens in the muds.

QUESTION: Why is air circulation better or worse than mechanical mixing? Cost?

ANSWER: We can't say which is better or worse. In our particular case because we didn't choose a very good pump to test, the air system turned out to be a more efficient device. I am certainly not going to generalize; however, as I indicated, particularly at deep depths where compression of air might be a problem, it may not be advantageous at all.

As far as costs go, because our lake was so small, these were very inexpensive devices. We rented the air compressor for $100 a month, and being only about 22 kilowatts, operating costs per hour were very small. We found used diffusers around the laboratory, so they cost us nothing. The pump and engine we bought was a more expensive item, and we had to have a raft with it. The pump cost about $1,500, and the raft and trailer cost about $1,000. The gasoline engine was about 15 kw, so the cost, particularly per unit volume, is really quite small and, I think, a favorable feature. You have to spend some money, but not really too much.

QUESTION: Could you control or regulate freezing of lakes and have you tried heating air prior to bubbling and what effect, if any, would heated air have on the lake environment?

ANSWER: We didn't try to heat the air and I can't think of any particular advantage to that; you might get a slight warming of the water as the air passed through. Whether this would be worth it or not, I couldn't say. Before I get to the first part of the question, I would like to add that we didn't use the air bubbles for transferring oxygen—people often ask about transferring oxygen straight from the bubble to the water. We estimate there was some of this, but the bulk of the reaeration came by bringing the deaerated water to the surface where it could contact the atmosphere for reaeration. As far as ice melting—we didn't do any work in the winter—there are companies that make devices for this which have been very effective for ice melting, but we haven't gotten into that at all.

QUESTION: What effect has destratification on benthos organisms (midges), mosquitoes, and aquatic gnats?

ANSWER: Again, we haven't looked into this. We would like to presume that creating an aerobic environment over the mud would be beneficial to the bottom organisms. This is one reason the fisheries biologists usually assume that we have improved the fishery. As for the flying nuisance organisms, mosquitoes, etc., we received no complaints from the fishermen on this lake, but that's a very qualitative estimate.

QUESTION: What is the difference between oxygen introduction at lower levels and aeration mechanisms in destratification?

ANSWER: If you're referring to the fact that we are just pumping the water to the surface where it can naturally reaerate as opposed to actually attempting to add gaseous oxygen, either from air or pure oxygen to the lower levels, there is a difference. In one case you destroy the thermal profile and depend upon the atmosphere for reaeration, and in the other one you have to apply your energy to maximizing the solution of oxygen into water and you might maintain your thermal profile. So, if I understand the question properly, that would be my comment.

QUESTION: Does hardness reduction because of algal blooms mean reduction in total dissolved solids and what relative magnitude of this phenomenon have you observed or can you predict?

ANSWER: In our particular lakes, we didn't have sufficient hardness to precipitate, but there are, of course, natural calcium carbonate deposits that have been attributed to the gross raising of pH by algal action. You can get a pH of 9 or higher under algal bloom conditions, and under those circumstances, you will get precipitation of calcium carbonate, something that would only happen in specialized cases, however.

SUMMARY AND CONCLUSIONS

Data acquired during this first phase of the long-range research project on the behavior of impounded waters and the effect of impoundment discharges on downstream water quality has dealt with the behavior of three water quality parameters in reservoirs; DO, synthetic organic materials, and nitrogen. In addition, a 3-year field study investigated the use of artificial destratification of reservoirs as an engineering method for maintaining good quality water throughout a reservoir. To present an overall summary of the entire 5-year research project, the major conclusions from each study are presented below.

General

1. More research is needed if the behavior of reservoir water quality is to be related to the environment, thereby permitting predictions of water quality in any given reservoir.

20. Artificial destratification has been used successfully by 11 water suppliers around the world as an effective technique for maintaining good raw water quality, and at this time (1969) is being considered for use by 9 others.

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Author's Note: To help the reader use this index, the following six items are presented. One, except as otherwise noted, all entries are in relation to impoundments. Two, the term ''impoundment'' refers to reservoirs, dammed lakes, and natural and artificial lakes. Three, most measurement entries are in the ''Analytical Methods and Techniques'' section. Four, the behavior of specific water quality constituents are listed under the constituent. Five, the terms ''destratification efficiency,'' ''oxygenation capacity,'' ''oxygen balance,'' and ''oxygen budget'' are used as defined in the text. Six, the only authors indexed are those whose written discussions appear in the text.

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(M.G.C.)

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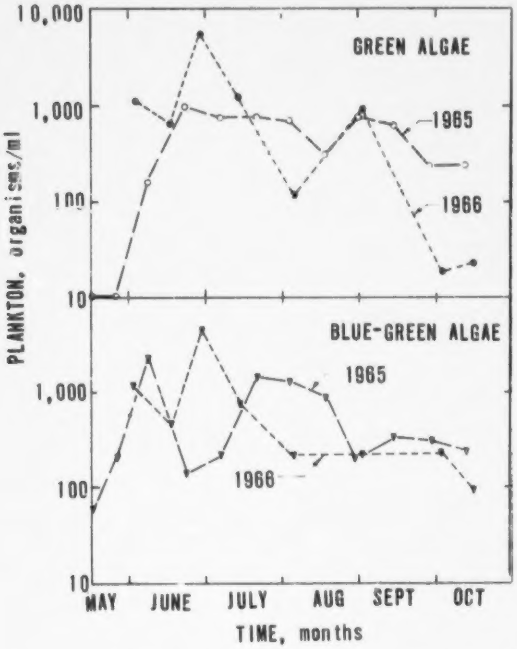

Figure 30. Green and blue-green plankton counts at the 5-ft depth in the unmixed lake.

reservoir. Blue-green algae cause the major taste and odor problems for water supplies, and a shift towards a population of green algae would be beneficial.

Evaluation of Equipment Performance

Because mechanical equipment is used for artificial destratification, some method is needed to design the most effective apparatus. Calculation of the ''oxygenation capacity'' and ''destratification efficiency'' as defined by Equations (4) and (5) is recommended.

$$\text{Oxygenation Capacity (OC)} = \frac{\text{Net Change in Oxygen Balance from } t_1 \text{ to } t_2}{\text{Total Energy Input from } t_1 \text{ to } t_2} \tag{4}$$

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DO Budget

2. The DO probe method using light- and dark-plastic boxes is a useful method for evaluating the DO budget in a deep reservoir where benthos and sedimentation do not affect the DO budget significantly.

3. The mathematical model:

$$D = \frac{k_1(L_\alpha)}{k_2-k_1}[\exp(-k_1 t) - \exp(-k_2 t)] + \frac{(R-P)}{k_2}[1-\exp(-k_2 t)]$$

$+ D_\alpha [\exp(-k_2 t)]$ is a promising candidate for describing the DO budget in reservoir waters.

4. The mathematical model can be solved for the five constants using the special three-DO-probe system and the computer program developed in this study.

Nitrogen Cycle

5. A nitrogen balance was attained in a clean, aerobic system by measuring only three of nine possible valence states of nitrogen, using an automated procedure for nitrogen analyses.

6. In the range studied, the initial ammonia nitrogen concentration and the initial nitrate nitrogen concentration had no effect on the initial rate of ammonia nitrogen oxidation.

7. In the range studied, the initial rate of ammonia nitrogen oxidation was proportional to the initial nitrifying seed concentration and the turbulence in the experimental apparatus.

8. In the presence of organics, if the initial ammonia nitrogen was not completely assimilated by the heterotrophic population during organic oxidation, simultaneous oxidation of carbonaceous and nitrogenous matter took place without influencing the initial rate of nitrification.

9. The presence of 5 and 10 milligrams of nitrite nitrogen per liter of water caused the oxidation of ammonia nitrogen to slow when introduced at the beginning of the test, but not when naturally produced from the oxidation of ammonia nitrogen.

Synthetic Organic Materials

10. Low temperatures reduced the rate of biodegradation of sodium lauryl sulfate (SLS), linear alkylate sulfonate (LAS), and 2,4-Dichlorophenoxy Na Acetate (2,4-D).

11. Low oxygen concentrations reduced the rate of biodegradation of SLS, LAS, and 2,4-D *more* than did low temperature.

12. The greatest resistance to biodegradation for the three synthetic organics studied was caused by the combined environmental effects of low temperature and low DO concentrations.

Artificial Destratification

13. Artificial destratification with some mechanical device is an effective method of improving water quality in reservoirs or maintaining good quality water in reservoirs, or both.

14. Artificial destratification creates a uniform temperature throughout the reservoir depth, adds DO to the water, oxidizes reduced materials such as sulfide and reduced iron and manganese, or prevents their formation, but does not impair the clarity of the water.

15. Although some nitrogen and phosphorus may be brought to the surface during artificial destratification, increased algal populations do not occur. Actually, because of some environmental change, the algal populations temporarily declined during artificial destratification.

16. During the decline in algal populations noted above, the blue-green algae population usually declined more than the green algae population, on a relative basis.

17. The decline in plankton during artificial destratification was only temporary, as the populations rose again after mixing stopped, but not to bloom levels.

18. Periodic artificial destratification throughout the summer season, starting in the spring, was a better method of water quality management than a single mixing in mid or late summer.

19. Calculation of the destratification efficiency (DE) and the oxygenation capacity (OC), as defined herein, is a useful method of comparing the mechanical performance of various artificial destratification devices and may be useful parameters for design.

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